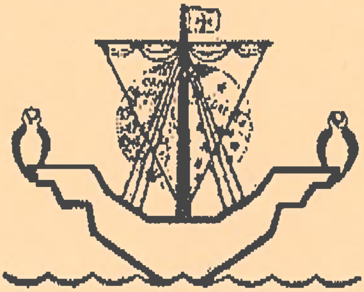
PO ISSN 0034-7736

REVISTA DE BIOLOGIA



EDITOR

MUSEU E JARDIM BOTÂNICO (M. N. H. N.)

VOL. 20 2002 N.^{os} 1-4

LISBOA

REVISTA DE BIOLOGIA

VOL. 20 2002 N.^{OS} 1-4

Publicação do
Museu e Jardim Botânico
Museu Nacional de História Natural
Universidade de Lisboa
Rua da Escola Politécnica 58, 1250-102 Lisboa (Portugal)

Esta Revista tem revisão científica.
Título para citações bibliográficas: *Revista Biol. (Lisboa)*

LISBOA

6°ENCONTRO NACIONAL DE ECOLOGIA

Promovido pela
SPECO - Sociedade Portuguesa de Ecologia

Lisboa, Novembro de 2001

Publicação financiada pela:

FCT Fundação para a Ciência e a Tecnologia
MINISTÉRIO DA CIÊNCIA E DA TECNOLOGIA

Montagem e arranjo gráfico no Museu e Jardim Botânico
por J. Cardoso, em Dezembro de 2002

PREFÁCIO

O 6º Encontro de Ecologia, realizado em Lisboa em Novembro de 2001, teve como tema principal a 'Fragmentação de Habitats e Populações', área que tem vindo a adquirir uma importância crescente no contexto da conservação da biodiversidade e do desenvolvimento sustentável. Sendo hoje inequívoco que a actividade humana tem promovido a perda da biodiversidade e alterado a paisagem a um ritmo nunca antes observado, afigurou-se como fundamental o debate sobre os efeitos deste processo nos diferentes níveis de organização da vida (genes, populações, comunidades e ecossitemas).

O presente volume da Revista de Biologia reúne alguns dos trabalhos que foram apresentados no âmbito do 6º ENE, e que se centraram nas principais questões subjacentes ao tema em discussão e que são, respectivamente, a identificação do problema (A Fragmentação da Natureza / A Natureza da Fragmentação), os efeitos do mesmo (Respostas Populacionais) e a procura de respostas adequadas (À Procura de Soluções).

Os artigos que constituem este número, para além das conferências proferidas pelos investigadores internacionais convidados, são contribuições originais de todos os que, tendo apresentado comunicação sobre a forma oral ou de painel, manifestaram interesse em ver publicado o seu trabalho. Os manuscritos submetidos para publicação foram objecto de revisão por parte de investigadores nacionais que fizeram críticas e sugestões, valorizando esta publicação de distribuição nacional e internacional.

Estamos certos de que este volume da Revista de Biologia, pela qualidade científica e técnica dos trabalhos publicados, contribuirá para a difusão do conhecimento e abrirá novos horizontes na investigação sobre Ecologia em Portugal.

A SPECO agradece o interesse de todos os que contribuíram com os seus trabalhos, aos que se disponibilizaram a rever e a sugerir alterações aos manuscritos, aos responsáveis pela edição da Revista de Biologia, Profª Paula Sobral, Drª Ireneia de Melo e Sr. José Cardoso, e às entidades que nos apoiaram e assim tornaram possível a sua edição.

A Direcção da
Sociedade Portuguesa de Ecologia

ÍNDICE

Palestras convidadas

Comunicações

LINKING THEORY, DATA, AND PRACTICE IN CONSERVATION: IMPLICATIONS FOR FRAGMENTATION STUDIES

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INTRODUCTION

Although the cumulative effects of widespread, unsustainable resource use were recognized over 150 years ago, the rudiments of a large scale science upon which to base consistent and effective management did not exist. Until recently, we assumed that smaller scale studies, employing small sampling units, could be scaled to provide answers. In the past two decades we have realized that scaling across hierarchical levels and landscape extents often results in qualitatively different patterns, making meaningful interpretation of data difficult or impossible (O'NEILL *et al.* 1985). Even at smaller scales, a close matching of theory with data is difficult. FAGERSTRÖM (1987) and HAILA (1988) argued that even the simplest ecological statements include more than can be concluded by observation alone. Additionally, indirect effects, for example the presence of trophic cascades (WOOTTON, 1994; but see STRONG, 1992 and CHASE, 2000 for counter arguments), as well as pulsed resources (OSTFELD & KEESING 2000), i.e., the availability of much higher than normal resources for short periods of time (mast crops) seriously complicate our understanding of species interactions, even when larger scale environmental constraints (the spatial explicitness of landscape pattern), and extreme stochastic weather events and other natural and anthropogenic disturbances are not considered. When landscape fragmentation and disturbance regimes are considered, spatial and temporal effects further complicate our attempts to understand species responses. Making a tight match then between theory and practice at any scale can become a formidable task and may not be possible, but an effective linking of relevant theory with management practice is not only possible, it is essential, if management is to be based on sound ecological principles.

In this paper, I address: a) the implications of a 'realist' philosophy of science by briefly examining the tenuous assumption of a direct correspondence between observation (data) and ecological reality; b) the problems posed by multi-meaning normative paradigms; and c) the putative causes of technical difficulties with the science. I then relate these ideas to the general issue of habitat fragmentation by evaluating a report that has assessed the efficacy of fragmentation experiments.

IMPLICATIONS OF A REALIST PHILOSOPHY

Linking theory and data with practice, in other words, providing a conceptually sound data base upon which to base management decisions, is not an easy task, especially at larger spatial extents and temporal horizons. There is a tendency for ecologists to adhere to what T. F. H. Allen has called a realist philosophy, i.e., a widespread belief that research data correspond on a 1:1 basis with ecological reality. Even a cursory examination of the results of studies reported in any ecology journal will demonstrate that simple correspondence does not exist. There is seldom a 1:1 correspondence between any set of data and ecological reality. Ecological complexity at all scales tends to blur research results, with patterns that are more often than not equivocal, especially when results from several studies are compared (DEBINSKI & HOLT, 2000). Hence one is left to interpret study results, if not from a 'realist philosophy', then in the light of what are assumed to be realistic but generalized theoretical and conceptual frameworks.

NORMATIVE PARADIGMS AND TECHNICAL DIFFICULTIES

Linking theory to management at the landscape level is a worthy goal, but one not easily achieved. There are at least two main difficulties. One rests with the nature of what conservation (the practice) means, what it includes, and how its concepts are interpreted; the other rests with the complex nature of larger-scale ecology (the science).

Difficulties with normative paradigms

The conservation literature is rife with multi-meaning concepts that are used to guide conservation efforts. For example, the normative concept of ecosystem management, i.e., managing for "ecosystem health with commodity extraction as an ancillary goal" (CALLICOTT *et al.,* 1999, GRUMBINE, 1997), was institutionalized by the U. S. Forest Service in 1992 (Forest Ecosystem Management Assessment Team 1993), but there has been controversy regarding its definition (STANLEY, 1995, GRUMBINE, 1994, 1997). To provide clarification, CALLICOTT *et al.,* (1999) organized 'normative' or 'umbrella' (NOSS, 1995) conservation concepts into two philosophical camps based on whether *Homo sapiens* was included as part of nature (functionalism) or not (compositionalism). They argued that compositionalism included the conservation concepts of biological diversity (WILSON, 1992), biological integrity (ANGERMEIER & KARR, 1994), and ecological restoration (Society for Ecological Restoration, 1997); clearly concepts where *Homo sapiens* is not considered a part of nature, but rather an intrusion. Functionalism included normative concepts such as ecological services (CONSTANZA, 1991, DAILY *et al.,* 2000), ecological rehabilitation (MEFFE, 1995), ecological sustainability (CALLICOTT & MUMFORD, 1997), ecosystem health (CONSTANZA *et al.,* 1992), ecosystem management (GRUMBINE, 1994, 1997), and adaptive management (WALTERS, 1986); concepts that attempt to harmonize anthropogenic influence and disturbance with the natural world. CALLICOTT *et al.,*

(1999) emphasized that these were extremes on a *continuum* and were presented for the sake of clarification. WILLERS (2000) and HUNTER (2000) criticized aspects of the dichotomy that CALLICOTT *et al.* (1999) presented, but it seems apparent that precise, clear, and standardized definitions of conservation concepts are helpful, regardless of where they lay on the *continuum*. When concepts are used loosely, much of their power is lost. Unless clear definitions are developed and used, conservation concepts become pseudo-cognates; that is, each person who uses the term feels that everyone else shares her/his definition (BISSONETTE, 1997), when upon closer evaluation, it is evident that is not the case. A finer grasp of meaning and context is imperative. Further, a closer examination suggests that some concepts as currently defined, cannot be made operational. For example, effective monitoring of biological diversity over space and time requires attention to the 'what', 'why', and 'how' of data collection (YOCCOZ *et al.,* 2001). As such, measurable state variables are required. Non-quantitative state variables, such as 'ecosystem health' (YOCCOZ *et al.,* 2001) are difficult or impossible to measure and hence, monitor. CALLICOTT *et al.* (1999) provided context and a framework that placed anthropogenic disturbances and influences as the defining distinction between the two philosophical camps. This would appear to be useful when addressing conservation conflicts and problems. It is especially important when we try to understand the effects of habitat fragmentation. Knowing the nature of the disturbance regime (abiotic *vs.* anthropogenic) that results in fragmentation allows one to assess the usefulness, appropriateness, and applicability of disturbance metrics used to describe the landscape. Context is everything. For example, descriptors such as frequency (the mean number of events per time period), return interval (the inverse of frequency or the mean time between disturbances) and predictability (a scaled inverse function of variance in the return interval) (WHITE & PICKETT, 1985) will have variable usefulness depending on whether the disturbance is abiotic and governed by a deterministic chaotic or similar dynamic, than if anthropogenic activities are the primary cause of disturbance.

Difficulties with the science

There has been considerable development in landscape theory in the past two decades and much discussion about how to deal with scale problems. We know what most of the relevant problems with the science are; we are only now learning how to solve them. Briefly, the science is almost always multi-causal (BISSONETTE, 1997), non-linear dynamics most often predominate (KAWATA, 1995), thresholds seem to exist everywhere (LEVIN, 1992, 1999), and scaling from one hierarchical level to another often leads to qualitatively different results (O'NEILL *et al.,* 1985). These characteristics result in such complexity of causal influences and constraints at multiple hierarchical levels and with contingent feedbacks, that many consider ecological systems to behave as complex adaptive systems (LEVIN, 1999). Understanding ecological complexity and applying that knowledge to solve pressing problems in conservation is the challenge. We have had limited success. Much remains to be done. A quest for understanding complexity remains the focus of much of the discipline of landscape ecology.

Ecologists in general have tended to ask 'solvable' questions. Messy, intractable, and complex problems have not been addressed until advances in technology or paradigmatic theory made them more 'tractable'. For example, the development of landscape metric software, the ability to gather and analyze digital landscape data, and the realization that temporal and spatial scale effects matter, have allowed ecologists to ask questions regarding the effects of landscape fragmentation that could not have been addressed earlier. However, even the most sophisticated technological tools are just tools, and new paradigms are always more complex than their initial use would indicate. Given these tendencies, it does not take much for ecologists to be insufficiently introspective when interpreting their results. This may even hold for the most rigorous scientific approaches. Many if not most ecologists would tend to agree that experimentation would provide more enlightenment to fragmentation problems than the more traditional observational, historical, and comparative approaches. DEBINSKI & HOLT's (2000) recent survey of habitat fragmentation experiments provides a troublesome message and suggests "not necessarily".

FRAGMENTATION EXPERIMENTS: WHAT IS THE MESSAGE AND IS IT CLEAR?

In an enlightening review of habitat fragmentation experiments, DEBINSKI & HOLT (2000) not only have provided an excellent analyses of the few studies that have actually used experiments to refute or support a priori hypotheses about fragmentation effects, but their results speak volumes about the complexity of causal mechanisms attendant to habitat change. DEBINSKI & HOLT (2000) reported the results of 20 studies (21 if a notational account given in the acknowledgements is included) that tested six major groups of hypotheses related to: 1) the relationship of species richness to area, or 2) species abundance or density to area, 3) whether interspecific interactions are modified by fragmentation, 4) whether edge effects influence ecosystem 'services', 5) the relationship between corridors and movement between habitat fragments, and 6) whether connectivity increases species richness. The results are interesting and informative. Only 6 of 14 (37.5%) studies supported the expectation of hypothesis 1 and 6, i.e., that species richness should increase with increasing area or connectivity. The expectation for hypothesis 2 is that specialist species abundance and density should decrease with increasing area, although movement dynamics between different habitat elements (matrix *vs.* patch) might result in increases. DEBINSKI & HOLT (2000) found that species abundance decreased with increasing fragmentation in only 6 of 13 studies (46.2%). The prediction for hypothesis 3 is that some modification is expected. This prediction was supported in the two studies that addressed the question. Significantly, both involved arthropod populations, suggesting that the authors made their measurements at the correct scale. An expectation from theory is that an increase in edge relative to core areas (hypothesis 4) can have profound effects on ecological processes (SAUNDERS *et al.*, 1991, DEBINSKI & HOLT 2000). Two of three (66.6%) studies supported the hypothesis on edge effects. The expectation for hypothesis 5 is that fragmentation inhibits movement and the presence of corridors, and hence connectivity should be positively correlated with increased movement. DEBINSKI & HOLT (2000) found that 4 of 5 (80%) of the studies they evaluated supported the hypothesis, at least for some species. They found mixed results for predictions involving species richness (e.g., LAURANCE & BIERRGAARD, 1996, COLLINGE, 1995 *vs.* SCHMIEGELOW *et al.*, 1997, MARGULES, 1996), density and abundance of species (e.g., FOSTER & GAINES, 1991, MARGULES & MILKOVITS, 1994 *vs.* BARRETT *et al.*, 1995, COLLINS & BARRETT, 1997), edge effects (e.g., BIERRGAARD *et al.*, 1992, KLENNER & HUGGARD, 1997 *vs.* ROBINSON *et al.*, 1992), and corridors and movements (e.g., HADDAD, 1997, WOLFF *et al.*, 1997 *vs.* Andreassen *et al.*, 1998). In their words, the results were "entirely mixed". DEBINSKI & HOLT (2000) credit time lags, scale inconsistencies, contingent social interactions between species, and habitat generalists as some of the causes of the results.

The hypotheses examined are of fundamental interest to ecologists who strive to understand the underlying conceptual framework governing fragmentation effects, and to managers charged with conducting 'ecosystem management' (WALTERS, 1986, WALTERS & HOLLING, 1990, Forest Ecosystem Management Assessment Team, 1993, GRUMBINE, 1994, 1997), or with conserving biodiversity (NOSS *et al.*, 1994, 1995, IUCN, 2000). The clear message appears to be that straight-forward predictions do not capture the multi-causal nature of organism response to fragmentation. For example, the distribution of a species may be related to different habitat features at different spatial scales, and generalist species can be expected to respond differently than specialists. Similarly, early successional species, transient (usually younger age individuals) organisms, and edge and core-sensitive species can be expected to respond differently to landscape fragmentation (BISSONETTE *et al.* 1997, STORCH, 2002, DEBINSKI & HOLT 2000). Time lags can be expected to be common and to have profound effects. Additionally, matching time and spatial domains remains a problem.

TEMPORAL AND SPATIAL EFFECTS

DELCOURT *et al.* (1983), WIENS (1989), HOLLING (1992) and BISSONETTE (1997) have all suggested that there is an approximate matching between temporal and spatial scales in ecological processes. For example, the global distribution of forested landscapes changes at a speed of tens of thousands of years; regionally, forest disturbances such as fire occur at intervals from a few to hundreds of years; and locally, vegetation changes within annual cycles. What this means is that one can expect time lags to be prevalent in landscape scale interactions. The decline of sugar maple (Acer saccharum) forests in the Northeastern United States over the past half century or more is an excellent example where not only is the spatial extent of the damage more difficult to quantify, but its cause exceedingly difficult to determine because of the time lags involved. Interestingly, the pattern of decline extends over hundreds of miles, but appears to be caused by localized soil acidification effects. Air-borne pollutants from the mid-western United States are carried by the prevailing west to east weather pattern and acidified pollutants are deposited in the eastern United States. Several hypotheses have been suggested for the large scale decline of sugar maple trees. Insect defoliation, drought, and historic land use practices have been proposed as causal (DRISCOLL *et al.,* 2001), but have not been strongly supported (SWISTOCK *et al.,* 1999, HORSLEY *et al.,* 2000, DROHAN, 2000). RATHER, SHARPE & SUNDERLAND (1995) and SHARPE *et al.,* (1999) argued persuasively that acid deposition on forest soils, coupled with very long lag times in soil response, account for the sugar maple decline. The mechanism appears to be soil acidification and is apparently referenced by critically low calcium-to-aluminum and calcium-to-magnesium ratios (SHARPE *et al.,* 1999, SWISTOCK *et al.,* 1999). Time lags of many years and perhaps even decades appear to be operating, suggesting that critical thresholds of soil acidity need to be reached before effects become apparent. This is but one example, however time lags and threshold effects may be more prevalent than ecologists have supposed. Much longer multi-scale studies are needed to understand the putative causes. Time lags also involve major ramifications for management. Land management or species conservation measures that are likely to take years to show the desired effects are difficult to defend and are not popular with decision makers, who work with annual budgets and may be elected for short periods of time. Additionally, the public and media appear to respond much more readily to more concrete and immediate results that appear to have a connection with causality closer to 1:1.

In their study, DEBINSKI & HOLT (2000) have done landscape ecologists a remarkable service. Their results tell us something very important about the effects of fragmentation. The essence of the message is that the effects of fragmentation can be understood as multi-causal, exhibiting thresholds where they are unexpected, are characterized by time lags that may be unpredictable, are heavily influenced by the structural differences between the matrix and the patches, especially if the patches are disturbance rather than remnant patches, are heavily dependent upon the temporal and spatial scales of observation, and whose dynamics are contingent on system history and therefore subject to unpredictable stochastic events. Importantly, species characteristics influence their responses to this myriad of causes and constraints. What we have just described is a complex adaptive system (LEVIN, 1999) that may be characterized by deterministic chaotic events (PEAK, 1997). Weather pattern dynamics are an excellent example of this kind of system. The question can be asked: do we think that we, as landscape-oriented animal ecologists, can beat the weatherman in prediction? Weather forecasts are notoriously poor once they exceed time frames of more than a few days, despite the enormous amounts of data and sophisticated analytical tools that meteorologists have at their disposal. Sensitivity to initial conditions, i.e., system history, has an enormous effect. So what makes us believe that we should be able to do much better? Perhaps the message is that when we seek specifics, they may always be local and empirically-based, making accurate prediction and scaling exceedingly difficult or impossible. However, at some general level of explanation, we should be able to manage an effective linking of relevant theory with management practice. To the extent that we can do this, the science is advanced and management becomes databased, giving us hope that what we do on the ground is reasonable and prudent.

ACKNOWLEDGEMENTS

I thank Dr. Margarida Santos Reis for her gracious invitation to present the opening keynote address at the 6th Annual Meeting of the Portuguese Ecological Society in Lisbon on 1 November 2001. I also thank Dr. I. Storch for her insight and helpful reading of the original material from which this paper was prepared. This contribution has been adapted from the introduction and conclusion chapters of an edited book, titled ***Landscape Ecology and Resource Management: Linking theory with Practice*** by J. A. Bissonette and I. Storch, and published by the Island Press in 2002. I also thank my agency, the U.S. Geological Survey for making this work possible. The Utah Cooperative Fish and Wildlife Research Unit is coordinated by the U.S. Geological Survey, Utah State University, the Utah Division of Wildlife Resources, the Wildlife Management Institute, and the U. S. Fish and Wildlife Service.

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MOLECULAR GENETICS OF FRAGMENTED POPULATIONS: CAN WE TRACK RAPID DEMOGRAPHIC CHANGE?

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INTRODUCTION

Over the last ten years it has become apparent that molecular markers can be used to inform and prioritise populations of threatened species for conservation (FRANKHAM *et al.,* 2002). However, it has also become apparent that fragmented populations pose special problems for population geneticists and that conventional analysis of allele frequency differences among populations can be at best misleading, due to the extremely strong effects of demographic changes leading to genetic drift, inbreeding and selection (e.g. O'RYAN *et al.*, 1998; HEDRICK (1999). These problems demand closer attention if conservationists are really going to depend on molecular genetic data and use these data to inform practical conservation policy.

In this brief essay I hope to highlight some of these problems using examples from the work of our laboratory. These problems fall into the categories of: a) using phylogenetic approaches and in particular the Evolutionary Significant Unit (ESU) and Management Unit (MU) to prioritise fragmented populations for conservation, b) inferring the significance of modern-day genetic diversity in the absence of historic specimens and c) detecting the severity (or indeed existence) of demographic bottlenecks in fragmented populations. These form some of the key elements of threatened species management given the fact that so many populations of endangered species are found in fragments that prioritisation will become a fact of life, that the requirement for the maintenance of genetic diversity in small populations is under constant question and because small population biological models tend to involve single sub-populations, a situation which is unlikely to hold for the majority of vagile species.

THE ESU/MU PROBLEM…

The relationship between phylogenetic units for conservation such as the ESU (Evolutionary Significant Unit; RYDER, 1986) and the MU (Management Unit; MORITZ, 1994b) and traditional taxonomic descriptors such as subspecies, races and ecotypes is at best obscure. Many authors have in the past equated the ESU level with that of subspecies (e.g. BALL & AVISE, 1992) an approach partly due to the original motivation for the ESU idea (RYDER, 1986). However, ESUs, by many definitions, are regarded as phylogenetic species and yet in some cases this could, in principle, be applied to very recently derived populations of the same species (e.g. TARR & FLEISCHER, 1999). The concept of the management unit (MU) is now commonly in use to diagnose subdivided populations where divergence time has not been sufficient to accumulate evolutionarily distinct characters or some other factor, such as limited gene-flow has kept the populations genetically non-independent (MORITZ, 1994b).

A common problem concerns the management of recently isolated populations, which may not, in many cases, be appropriately classed as ESUs. Such populations are sometimes demographically inviable and may possess low amounts of genetic variation. In these cases the MU may be a more applicable approach (e.g. BRITTEN *et al.,* 1997; BAKER *et al.,* 1998; O'RYAN *et al.,* 1998). Many of these studies are being carried out with the aim of identifying management units for translocating individuals to augment demographically inviable populations (MORITZ, 1999). A major criterion identified by MORITZ (1994b) for defining separate management units is the possession of significant haplotype frequency differences at mitochondrial DNA (not necessarily at nuclear loci) regardless of the phylogenetic distinctiveness of the alleles.

Populations that have undergone rapid and radical changes in their habitat quality and quantity, degree of fragmentation and isolation from other populations pose special problems when interpreting genetic data and assignment of MU status needs to be carried out with caution. For example, the extreme demographic fluctuations which may be relatively common in small isolated populations is likely to result in genetic drift and/or inbreeding, thereby accentuating allele frequency differences and result in the further loss of alleles (e.g. SACCHERI *et al.*, 1998, 1999) and may result in the fixation of alleles which could be locally unique. It is common for isolated populations to possess one or two mitochondrial alleles, and many such populations may have suffered serious decline during the last 200 years. Sampling of these populations may further lead to apparent differentiation among populations (e.g. SJÖGREN & WYÖNI, 1994). The genetic patterns often observed in endangered populations therefore result from recent demography as opposed to long-term divergence, potentially complicating translocation plans.

As an example, BARRATT *et al.,* (1999) found a large number of mitochondrial haplotypes, some of which were highly divergent, in small, isolated populations of the red squirrel in the UK. The frequencies of these alleles were also extremely different, with many populations only containing alleles unique to the dataset (Tab. 1). However, phylogenetic analysis of these sequences revealed no consistent pattern of diversity among haplotypes in different populations, either in the UK or Europe. However, the red squirrel is known to have been extremely common, widespread and continuously distributed across Eurasia before deforestation for agriculture in the middle ages, and has been decimated in the UK and now southern Italy following the introduction of the American grey squirrel in the 19th century. As a consequence many southern UK populations that are today small and isolated have only been threatened for a few hundred years and may have indeed exchanged genes with neighbouring mainland European populations prior to the flooding of the English Channel 9,000 BP.

Under the criterion of minimum diagnosability in phylogenetic conservation, where possession of an unique haplotype is an irreducible unit of conservation (VOGLER & DESALLE, 1994) the red squirrel has many diagnosable ESUs (indeed, on might argue, many phylogenetic 'species') in the mitochondrial data set, and certainly each population would be considered a separate management

Table 1: Genotype diversity in UK and continental Europe populations of the red squirrel (adapted from Barratt *et al.*, 1999 for further details. General descriptors: B = Belgium; CI = English Channel Islands; E = England; G = Germany; S = Scotland; W = Wales).

Type	Fife (S)	Lanc (E)	The (E)	Arg (S)	Cum (E)	Spa (E)	Kie (E)	Arr (S)	Tor (S)	Jer (CI)	IoW (CI)	Clo (W)	Hal (G)	Bav (G)	Bel (B)
c17	-	-	0.23	-	-	-	-	-	0.875	-	-	-	-	-	-
h9	0.90	-	-	-	-	-	-	-	-	-	-	-	-	-	-
arg1	-	-	-	0.125	-	-	-	-	-	-	-	-	-	-	-
ahl78	-	-	-	0.50	1.0	0.06	-	-	-	-	-	-	-	-	-
f324	-	-	-	-	-	0.06	-	-	-	-	-	-	-	-	-
ahl41	-	-	-	0.125	-	0.44	1.00	-	-	-	-	-	-	-	-
f303	-	-	-	-	-	0.38	-	-	-	-	-	-	-	-	-
ahl81	-	-	-	0.125	-	-	-	-	-	-	-	-	-	-	-
wc3	-	-	-	-	-	-	-	-	-	-	-	0.18	-	-	-
iw3	-	-	-	-	-	-	-	-	-	-	0.29	-	-	-	-
j12	-	-	-	-	-	-	-	-	-	0.82	-	-	-	-	-
s6	-	-	-	-	-	-	-	0.50	-	0.18	-	-	-	-	1.0
g6	-	-	-	-	-	-	-	-	-	-	-	-	1.0	-	-
g8	-	-	-	-	-	-	-	-	-	-	-	-	-	0.20	-
g13	-	-	-	-	-	-	-	-	-	-	-	-	-	0.10	-
g15	-	-	-	-	-	-	-	-	-	-	-	-	-	0.10	-
a2	-	-	-	-	-	-	-	0.50	-	-	-	-	-	-	-
ahl35	-	0.15	-	0.125	-	-	-	-	0.125	-	-	-	-	-	-
g7	-	-	-	-	-	-	-	-	-	-	-	-	-	0.30	-
g11	-	-	-	-	-	-	-	-	-	-	-	-	-	0.20	-
g14	-	-	-	-	-	-	-	-	-	-	-	-	-	0.10	-
wc9	-	-	-	-	-	-	-	-	-	-	-	0.82	-	-	-
iw6	-	-	-	-	-	-	-	-	-	-	0.71	-	-	-	-
ahl63	0.10	0.85		-	-	-	-	-	-	-	-	-	-	-	-
m14	-	-	-	-	-	0.06	-	-	-	-	-	-	-	-	-

unit under standard criteria. However in the absence of phylogenetic structure in the populations analysed, and with the strong possibility of a purely demographic explanation of the data, it is important to consider whether any of the populations even represent separate management units. For populations with large numbers of alleles, such alleles may be found due to long term population stability, rapid generation time and/or large effective population sizes. In these cases, population fragmentation and the sub-sampling of a diverse mitochondrial gene pool could immediately produce significant allele frequency differences among populations (e.g. CORNUET & LUIKART, 1996), a pattern that potentially carries little or no evolutionary signal.

Therefore, under the circumstances described above for small subpopulations which are essentially remnants of once large, continuous and diverse populations, significant allele frequency differences or even fixation of different alleles in mtDNA can, in principle, accumulate in relatively few generations. Further, because of the small amounts of time since the fragmentation of many populations, population augmentation through translocation would be expected to be unlikely to result in genetic incompatibility. This does not preclude the possibility that locally adapted phenotypic characters may have become fixed within smaller isolated populations, a potential problem that may sometimes be tested for (CRANDALL *et al.*, 2000). The role of demography as a complicating factor in designating genetic management units is potentially important (and, for example, it predominates in domesticated species), and its incorporation into criteria setting on a case-by-case basis is a necessity.

THE PRESENT-DAY DIVERSITY PROBLEM…

The role of demographic and environmental stochasticity in population dynamics and persistence has been intensively studied by population ecologists since the 1960's, and has thus gained much credence as a result. Demographic management of critically endangered populations either *in situ* or *ex situ* is known to be of fundamental importance for their survival. More controversial, however, is the role that genetic variation and its loss through drift and inbreeding have in population persistence, since although this issue has received much attention over the last 20 years, documented examples of the importance of molecular homozygosity to population survival are sparse. However, the recent study of SACCHERI *et al.*, (1998) linking heterozygosity with probability of subpopulation extinction in a metapopulation of Glanville fritillary butterflies, followed closely by a study demonstrating that augmenting genetic variation in a threatened snake population reversed a long-term decline (MADSEN *et al.*, 1999) have revitalised the debate. However, the need to manage genetic diversity in species that live in small populations naturally remains controversial. At the level of the individual, studies of inbreeding and fitness have also been relatively sparse, however, the last ten years have seen the publication of several extremely important studies of inbreeding (and inbreeding depression), both for wild and captive populations. Many of these studies have explicitly applied molecular approaches to measure genetic diversity at the individual level (quantified, for example, by heterozygosity), within pedigrees or within small populations. Unsurprisingly, many studies have concerned island endemic species, a group which has suffered disproportionately from the effects of human intervention, and a group which is also expected to be least resilient in the face of environmental, demographic and genetic fluctuations (FRANKHAM, 1997).

One of the most celebrated examples of a conservation success involving *ex situ* management in a bird is the Mauritius kestrel (*Falco punctatus*) that has recovered from a single wild breeding pair in 1974 to a wild population consisting of over 200 breeding pairs by 1990 (Figure 1a). However, the question predominating the conservation literature for the past five years, especially applicable for small endemic populations is whether such resilience is to be expected due to the gradual removal of deleterious genes over millennia by selection – a process known as 'purging', whether rare mutations may produce an even greater genetic load than found in naturally large populations or whether endemic populations retain genetic diversity in line with neutral expectations accordingly to their effective population size. The first hypothesis would predict a genetic resilience to population bottlenecks whereas the second would predict an increased susceptibility to genetic extinction. This question was explicitly addressed in the kestrel study.

GROOMBRIDGE *et al.,* (2000) measured the loss of genetic variation resulting from the bottleneck in the Mauritius kestrel using microsatellite markers typed for modern birds and museum skins up to 170 years old. Although extant individuals showed predicted low levels of diversity, variability in the museum specimens was remarkably high (Figure 1b) and when compared to the genetic diversity expected for continental kestrel species, was found to be of a similar magnitude for a species of its range and hence effective population size (consistent with hypothesis 3; see Figure 1c). Interestingly, therefore, the Mauritius kestrel did not survive because of unsuspected additional genes in the wild population or because of a reduction of its genetic load due to a history of small population size, inbreeding and drift. In fact allelic diversity fell by 55% and heterozygosity fell by 57% during the bottleneck, and allelic diversity probably fell by a much higher percentage since these estimates are much more sensitive to the limited sample available through museum specimens. The fact that this species recovered without augmentation suggests that it was only weakly affected by this bottleneck, probably by chance, due to the genetic load of the individuals passing through the bottleneck. The generality of this observation is, however, difficult to assess in the absence of temporal or spatial replicates. Its significance, especially given the fact that many island endemics remain in real threat of extinction, is that there may be no 'special case' for managing diversity in island endemics, and that although this population patently survived, many others may not without genetic management.

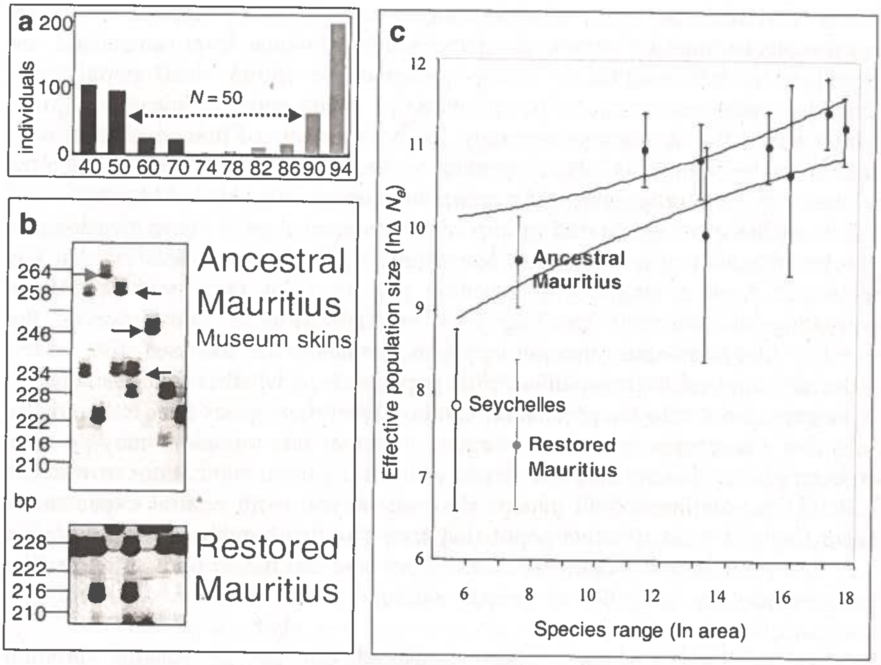

Figure 1a-c - Genetic diversity in ancestral Mauritius kestrels compared to modern continental kestrel populations; (a) demographic trajectory of the bottleneck; (b) genetic diversity using a microsatellite locus – arrowed bands are 'ghost' alleles unique to the ancestral samples; (c) relationship between relative effective population size and species range for a number of kestrel species (adapted from GROOMBRIDGE *et al.*, 2000).

THE DETECTING BOTTLENECKS PROBLEM…

The genetic trajectory of natural bottlenecks have rarely been documented, and in the (usual) absence of museum specimens, researchers are sometimes left with interpreting the signatures of such bottlenecks in the genes of present day populations (CORNUET & LUIKART, 1996), but how often can they detect them? The significance of the study of Mandarte Island song sparrows (*Melospiza melodia*) by KELLER *et al.* (1994) is that the population was followed through two bottlenecks in real time, so that real demographic data could be collected and, with the aid of a comprehensive pedigree of the island's small population, studbook estimates of inbreeding coefficients could be made. The consequence of this is that the genetic effects of a severe winter population crash, during which 95% of the population was killed, could be assessed. Inbred individuals were shown to have survived the crash much less well than non-inbred birds, and as a consequence this example has become one of the few convincing demonstrations of inbreeding depression in any wild population.

Subsequent analysis of the genetic variability dynamics of this bottleneck using molecular markers, (KELLER *et al.*, 2001) showed some striking results. Although, during the bottleneck, heterozygosity and allelic diversity were reduced close to neutral theory expectation, these measures regained pre-bottleneck levels within two years of the crash, much faster than expectation, so that a sample taken three years after the crash would show no evidence of it having occurred, although average inbreeding did increase rapidly over this period. Low levels of immigration (one female arrived immediately after the storm) and genetic drift account for this recovery. Figure 2 shows the effect that immigration had on expected heterozygosity. The descendents of crash survivors showed values reduced from 0.78 immediately prior to the crash to 0.64 five years after.

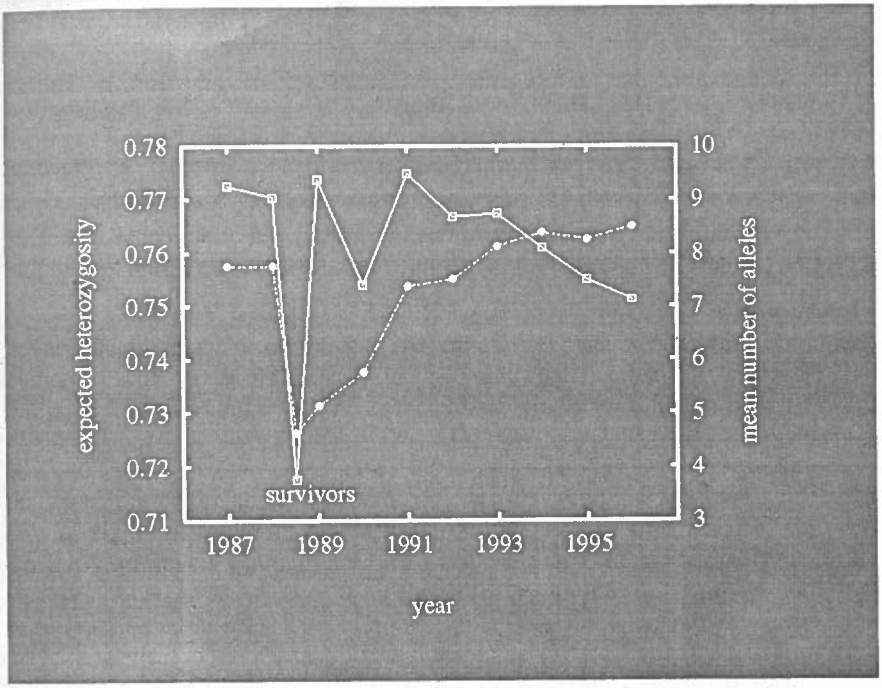

Figure 2 - The trajectory of average expected heterozygosity before, during and after the crash for all samples (open circles) and the subset excluding all immigrant lineages (closed squares). Adapted from KELLER *et al.*, 2001.

The role of inbreeding and drift in population and individual viability has been explored mainly in birds than invertebrates to date. As these examples and others show, much is still to be learned. More lessons need to be drawn from the many long-term studies of populations, both for endangered and stable species, and molecular studies on such populations are bound to be invaluable in highlighting new facets of these studies.

CONCLUSIONS

Studies of both naturally and anthropogenically fragmented populations of endangered species will dominate the conservation biology literature for decades to come. This unfortunate reality therefore begs the question as to what elements of genetic diversity we should be focusing on, and how should we bring molecular biology into routine use for practical conservation. There has been a tendency in the past for geneticists to look for generalities and guiding principles in the use of molecular data for describing units for conservation, assigning notional extinction risks on the basis of present-day genetic diversity and of applying to single population thinking to in practice much more complex systems. What is emerging from the literature steadily over the last few years is a bewildering variety of contexts, results and conclusions, which leads the author to the conclusion that single rules are inappropriate and indeed almost disrespectful of the complexity of evolutionary processes which have led to biodiversity on earth in 2002.

In this paper, through the study of red squirrels (BARRATT *et al.*, 1999) we have questioned the universal application of any form of phylogenetic species, ESU or MU definition in cases where lineage sorting is a direct consequence of recent and severe demographic contraction from a diverse gene pool, yet this is only one such example. In a recent study of island populations of the Komodo dragon from our laboratory, CIOFI *et al.*, (1999) found extreme genetic differentiation between the population on the island of Komodo and the other island where the species is found. Batymetric data indicated that Komodo was last joined to these islands during the glacial maximum 130,000 bp. Limited or a lack of gene flow between Komodo and the other islands subsequently may have accounted for this extreme differentiation. However, for several microsatellite loci analysed as part of this study, this divergence must have been due to mutation since allele sizes did not overlap among islands which Komodo having an aberrant distribution. There is therefore a strong *de facto* case on the basis of these data to regard dragons from the island of Komodo as a separate ESU. Yet there is no morphological differentiation among islands, and mitochondrial cytochrome *b* sequences were found to be monomorphic among all islands, suggesting a recent common evolutionary origin. Clearly, therefore, the use of multiple data sets and caution when interpreting single locus data are of paramount importance in such studies (in the case of the Komodo population we suggested management as a separate MU only).

Finally, the interpretation of modern day genetic diversity in conservation biology has received a huge amount of attention and criticism. Simply measuring expected heterozygosity in a threatened population and comparing this result with other populations or species studied perhaps at different times using different markers, and without paying attention to the historical processes which may have been implicated in present day patterns, will not allow ration genetic conservation to progress. Of clear benefit in this regard is the recent innovation of a suite of new analytical approaches, which are being developed which allow better inference of past population processes from present day phylogenetic and gene frequency data. In particular, the use of Markov chain Monte Carlo simulation in a genealogical (coalescent) framework has enabled conservation biologists to take a Bayesian approach to compare models of population history, given a set of molecular data (e.g. O'RYAN *et al.*, 1998; BEAUMONT, 1999; see BEAUMONT & BRUFORD, 1999 for a general description). These methods promise to significantly enhance our ability to assist in process-based genetic management by evaluating several models simultaneously, which might explain a given set of data – for example, evaluation of the relative roles of genetic drift versus gene flow (CIOFI *et al.*, 1999), assessing the timing and magnitude of admixture events (CHIKHI *et al.*, 2001) and cryptic hybridisation (BEAUMONT *et al.*, 2001) and assessing modern day diversity (e.g. GROOMBRIDGE *et al.*, 2000). Such analyses promise to revolutionise how we interpret genetic diversity in fragmented populations and incorporate a more process-based approach to conservation instead of trying to 'hold what we have' in a increasingly dynamic environment.

ACKNOWLEDGEMENTS

I would like to thank all my co-authors on the papers cited in this review for their help and collaboration. Especially I would like to thank Elizabeth Barratt, Mark Beaumont, Lounès Chikhi, Claudio Ciofi, Jim Groombridge, Lukas Keller, Richard Nichols and Ilik Saccheri, all of whose thoughts on the genetics of fragmented populations have profoundly influenced my own. This work was largely carried out in the Conservation Genetics Group at the Institute of Zoology, Zoological Society of London and I would like to thank all the other staff in the group and management at ZSL for both moral and financial encouragement. This work was supported by the Institute of Zoology, University of Florence, the Durrell Institute of Conservation and Ecology, Jersey Wildlife Preservation Trust, the Swiss National Science Foundation, the Natural Environment Research Council, The UK Ministry of Agriculture, Fisheries and Food (now DEFRA) and English Nature.

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CUSTOS ASSOCIADOS AO CRESCIMENTO DE FOLHAS E CAULES EM SOBREIRO E AZINHEIRA

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Oliveira, G. & Martins-Loução, M. A. (2002). Custos associados ao crescimento de folhas e caules em sobreiro e azinheira. *Revista Biol. (Lisboa)* **20**: 29-36.

Este trabalho integrou o estudo simultâneo da biomassa, morfologia, bioquímica e custos de formação de folhas e caules, para comparação do investimento energético em crescimento aéreo de duas espécies arbóreas: *Quercus suber* e *Quercus ilex* spp. *ballota*. As amostras foram recolhidas num montado misto de sobro e azinho no sul de Portugal. Registou-se grande variabilidade intra-específica na produção anual de biomassa (folhas e caules) em *Q. ilex*. As folhas e caules de azinheira eram mais ricos em celulose, hemicelulose e lenhina, e mais pobres em lípidos, proteínas e açúcares solúveis que as de sobreiro. Tais diferenças contribuiram para elevar significativamente os custos de formação, por unidade de massa, de *Q. suber*, o que contraria as teorias que associam menor longevidade a menores investimentos energéticos na formação dos órgãos. Observou-se que a produção anual de folhas e caules pode ser diferente em duas espécies filogeneticamente próximas e que partilham as mesmas condições ambientais, o que altera o esforço energético global de cada uma. Nesta perspectiva, uma das azinheiras estudadas apresentou maiores custos energéticos na formação de nova biomassa. Por outro lado, apresentando menores custos de formação por unidade de massa seca, poder-se-ia esperar que produzisse mais biomassa do que *Q. suber*. No entanto tal não foi sempre observado. Conclui-se que a comparação inter- e intra-específica não deve basear-se nos custos energéticos por unidade de massa mas sim nos custos globais da biomassa produzida.

Palavras-chave: biomassa, crescimento, custos de formação, longevidade foliar, *Quercus ilex*, *Quercus suber*.

Oliveira, G. & Martins-Loução, M. A. (2002). Growth and construction costs of leaves and stems of cork- and holm-oaks. *Revista Biol. (Lisboa)* **20**: 29-36.

Biomass is often used to evaluate the adaptive success of a species under given environmental conditions. However, it does not allow to adequately evaluate plant performance, because the same amount of biomass may imply differences in tissue construction, maintenance and/or repairment costs, in allocation of investment to organs, and in their longevity. The integrated study of phenology, production, and energetic costs involved (scarce in the bibliography) may help filling this gap. In the present work, a simultaneous study of biomass, biochemistry and construction costs was made on leaves and stems of two tree species - *Quercus suber* and *Quercus ilex* spp. *ballota*. The samples were collected by the end of a growth period, in a mixed stand of cork-oak and holm-oak in south Portugal. There was a great intra-specific variability in the annual production of biomass (leaves and stems) in *Q. ilex* because some trees produced two growth flushes. Holm-oak leaves and stems were richer in cellulose, hemicellulose and lignin, and poorer in lipids, proteins and soluble sugars than cork-oak. This contributed to increase the construction costs per unit dry weight of leaf in *Q. suber*. Greater leaf longevity of holm-oaks was therefore not associated with higher construction costs. The costs of annual production of leaves and stems were different in *Q. ilex* and *Q. suber* facing similar environmental conditions. Lower construction costs per unit dry weight in *Q. ilex* were not clearly related with higher aboveground biomass production. The results did not support the theories which link longer leaf longevity with higher construction costs, and highlight the need to consider the whole-biomass costs for adequate inter- or intra-specific comparisons.

Keywords: biomass, contruction costs, growth, leaf longevity, *Quercus ilex*, *Quercus suber*.

INTRODUÇÃO

A biomassa é um parâmetro frequentemente utilizado para avaliar o sucesso adaptativo de uma espécie em determinadas condições ambientais. No entanto, não permite avaliar correctamente o desempenho das plantas, pois a mesma quantidade de biomassa pode implicar diferenças importantes nos custos de formação, manutenção e/ou reparação, nos investimentos em termos de órgãos e na longevidade dos mesmos. O estudo integrado da fenologia, da produção e dos custos energéticos envolvidos (pouco frequente na bibliografia) poderia, pois, colmatar esta lacuna.

Foi objectivo deste trabalho avaliar o crescimento e respectivos custos energéticos de duas espécies próximas, *Quercus suber* e *Quercus ilex* spp. *ballota*, com as seguintes hipóteses subjacentes: i) os custos de formação da biomassa aérea são maiores em *Q. ilex*, atendendo à sua maior longevidade foliar (KIKUZAWA, 1995, POORTER & VILLAR, 1997); ii) há maior produção de biomassa aérea em *Q. suber*, atendendo aos menores custos de formação de folhas menos longevas (BARUCH & GOLDSTEIN, 1999).

MATERIAL E MÉTODOS

A amostragem decorreu no fim do período vegetativo, em Março de 1994, num montado misto de sobro e azinho perto de Alcáçovas (Évora). Recolheram-se seis ramos com mais de três anos em duas árvores adultas por espécie; de cada árvore amostraram-se três ramos expostos a Sul e três ramos expostos a Norte.

Cada ramo foi separado nos seus componentes: caules e folhas de 1 ano (produzidos na Primavera de 1993), 2 anos, etc. Determinou-se o comprimento e número de caules, e o número e área das respectivas folhas. Após secagem a 60°C durante 48h, este material foi pesado e, posteriormente, moído para quantificação de açúcares solúveis, amido, celulose, hemicelulose, lenhina, proteínas e lípidos. As amostras foram analisadas num espectrofotómetro de reflectância do infra-vermelho próximo (NIRSystems 6500) (JOFFRE *et al.*, 1992). Os custos de formação (CF, g glucose.g^{-1} matéria seca) calcularam-se com os coeficientes referidos na literatura (Tabela 1).

As comparações entre as duas espécies foram realizadas por ANOVAs a um factor para cada variável estudada, considerando-se significativas as diferenças com $p<0,05$.

Tabela 1. Custos de formação (CF) e função de diversos componentes do material vegetal (PENNING DE VRIES *et al.*, 1974, OECHEL & LAWRENCE, 1981, WILLIAMS *et al.*, 1987). O valor referente às proteínas assume o nitrato como fonte de azoto.

FRACÇÃO	**CF** (g glu.g^{-1} ms)	**FUNÇÃO ATRIBUÍDA**
Açúcares solúveis	1,09	metabolismo
Amido	1,17	metabolismo, armazenamento
Celulose	1,17	estrutura
Hemicelulose	1,17	estrutura
Lenhina	2,32	estrutura, protecção
Proteínas (NO_3^-)	2,48	metabolismo, armazenamento, protecção
Lípidos	3,03	metabolismo, armazenamento, protecção

RESULTADOS

Não se registaram diferenças significativas entre os ramos colhidos a Sul e a Norte, tratando-se, por isso, conjuntamente, os seis ramos de cada árvore amostrada. Registou-se grande variabilidade fenológica em *Q. ilex*, devida à ocorrência de dois períodos de crescimento consecutivos numa das árvores (Fig. 1). A ramificação foi mais intensa em *Q. ilex*, que ainda apresentava folhas de 2 anos, ao contrário de *Q. suber* (Fig. 1A). Os módulos anuais de sobreiro tinham maior biomassa (caules e folhas) (Fig. 1B) e área foliar (Fig. 1C) que as de azinheira (*Qi* 1), invertendo-se drasticamente esta relação no caso de *Qi* 2.

A composição bioquímica foi semelhante entre as árvores de cada espécie, pelo que se apresentam os valores médios (Fig. 2). As folhas de *Q. suber* eram significativamente mais ricas em açúcares, lípidos e proteínas que as de *Q. ilex*, sendo proporcionalmente mais pobres em celulose, hemicelulose e lenhina. Os caules de azinheira tinham significativamente maior concentração de celulose que os de sobreiro. Das diferenças na composição resultaram custos de formação significativamente mais elevados em *Q. suber*.

Em termos globais (Fig. 3), a produção de um módulo de crescimento anual foi mais cara em *Q. suber* do que em *Q. ilex* (*Qi* 1). No entanto, esta tendência alterou-se profundamente em *Qi* 2. Embora se saiba que segundos crescimentos podem ocorrer em ambas as espécies, não se conhecem as magnitudes dos mesmos nem os factores que os condicionam, pelo que não é possível determinar até que ponto foi excepcional o caso aqui estudado.

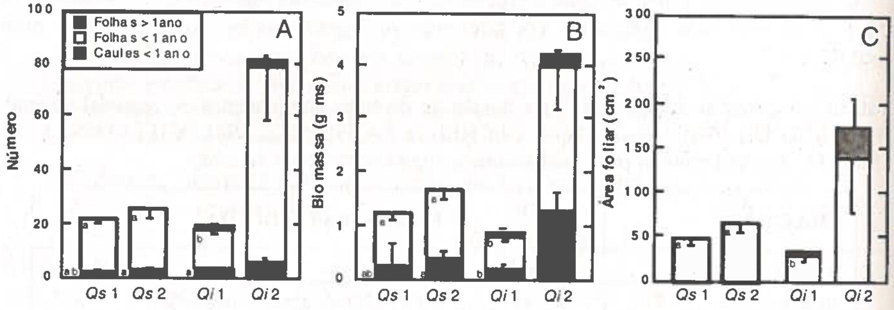

Figura 1. Produção de folhas e caules por módulo anual em *Q. suber* (*Qs*) e *Q. ilex* (*Qi*), em 1993. A. Número. B. Biomassa. C. Área foliar. Para *Q. ilex* indicam-se também valores referentes a folhas do ano anterior. As barras representam médias (± erro-padrão) de 6 ramos de cada árvore (*Qs* 1, *Qs* 2, *Qi* 1, *Qi* 2). Letras diferentes indicam diferenças significativas entre médias ($p<0,05$). Os valores de *Qi* 2 não foram incluídos na análise estatística por serem claramente diferentes dos restantes.

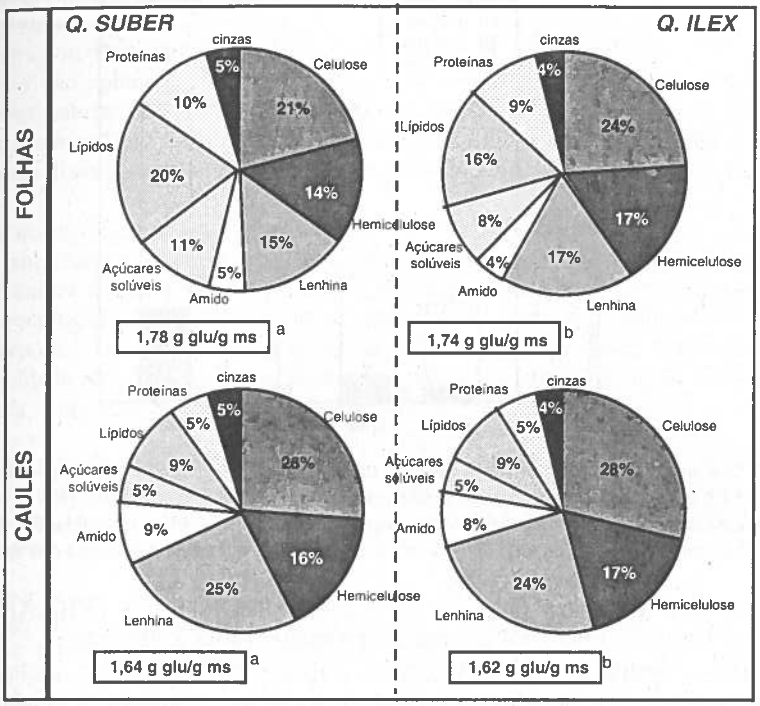

Figura 2. Composição das folhas e caules produzidos em 1993 por *Q. suber* e *Q. ilex*. Em cada caso indica-se o custo de formação por unidade de massa, e a significância das diferenças entre espécies (p<0,05). Os valores representam médias de 6 ramos e 2 árvores por espécie.

DISCUSSÃO

Os resultados indicaram que os custos de produção do módulo anual (folhas e caules) podem ser diferentes em duas espécies filogeneticamente próximas partilhando as mesmas condições ambientais. No entanto, não permitem aceitar as hipóteses iniciais, pois os custos de formação foram menores em *Q. ilex*, a espécie com maior longevidade foliar. Por outro lado, os maiores custos de formação de *Q. suber* não envolveram claramente menor produção de biomassa aérea relativamente à outra espécie. Estes resultados parecem contradizer as teorias que associam a maior longevidade a maiores custos de formação e/ou menor capacidade

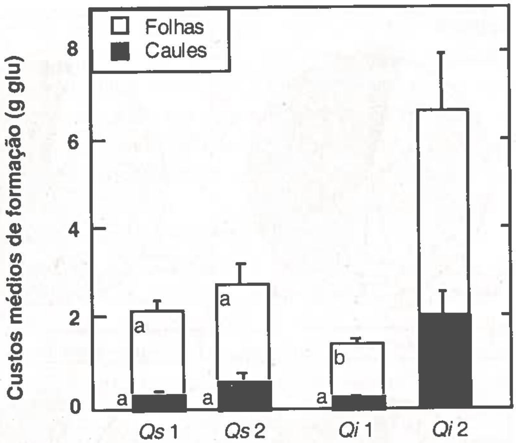

Figura 3. Custos médios de formação de um módulo anual de *Q. suber* (*Qs*) e *Q. ilex* (*Qi*) em 1993. As barras representam médias (± erro-padrão) de 6 ramos por árvore (*Qs* 1, *Qs* 2, *Qi* 1, *Qi* 2). Letras diferentes indicam diferenças significativas entre médias ($p<0,05$). Os valores de *Qi* 2 não foram incluídos na análise estatística por serem claramente diferentes dos restantes.

fotossintética da folha (KIKUZAWA, 1995, POORTER & VILLAR, 1997, DAMESIN *et al.*, 1998). Tal contradição poderá ter várias explicações:

1. Maior investimento em outros órgãos e tecidos em *Q. ilex*, não contabilizados neste estudo (e.g. raízes, troncos, frutos). De facto, a produção de bolota é maior em *Q. ilex* do que em *Q. suber* (IFN 2001). Isto levaria à diferente distribuição dos custos energéticos (GRIFFIN *et al.*, 1993, AERTS, 1995, STEVENSON & SHACKEL, 1998), diminuindo os relativos ao crescimento vegetativo aéreo (*Qi* 1). *Qi* 2 representaria assim um caso excepcional de baixo investimento em crescimento subterrâneo e/ou produção de frutos a favor da expansão da copa, pelo menos em 1993.
2. Maior capacidade fotossintética de *Q. ilex*, o que implicaria menor necessidade de incremento de área foliar. Mas isto implicaria grande variabilidade intra-específica na capacidade fotossintética, para explicar as diferenças entre as duas azinheiras. Por outro lado, os dados bibliográficos disponíveis sugerem poucas diferenças entre as duas espécies (GARCÍA-PLAZAOLA *et al.*, 1997, INFANTE *et al.*, 1999, MEDIAVILLA *et al.*, 2001).
3. Diferentes eficiências na utilização dos recursos (BARUCH & GOLDSTEIN, 1999) entre árvores e/ou entre espécies.

4. Maiores custos de manutenção (respiração) em *Q. ilex*, associados à maior longevidade foliar.
5. Amostragem demasiado reduzida.
6. Teorias não aplicáveis a espécies filogeneticamente próximas e que partilham o mesmo habitat. De facto, tais teorias baseiam-se na comparação de espécies contrastantes (e.g. herbáceas *vs* lenhosas, perenifólias *vs* caducifólias, etc.), de géneros distintos e/ou ecossistemas diferentes (POORTER & VILLAR, 1997).

Tanto quanto é do nosso conhecimento, este é o primeiro trabalho que apresenta estudos simultâneos de biomassa, composição bioquímica e custos de formação de folhas e caules em duas espécies lenhosas mediterrânicas. Embora os resultados não sejam conclusivos, apontam questões importantes para a discussão do valor adaptativo da relação entre custos de formação e longevidade foliar, bem como para a necessidade de considerar os custos energéticos a nível de toda a biomassa produzida, e não apenas por unidade de massa.

AGRADECIMENTOS

Trabalho realizado no âmbito do projecto MOST (EV 5V-CT92-0210). Agradecemos a todos os colegas que colaboraram no trabalho de campo e de laboratório. As análises foram graciosamente efectuadas por R. Joffre (CNRS – Montpellier).

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ALTERAÇÕES NAS COMUNIDADES BENTÓNICAS DE SUBSTRACTO MÓVEL PROVOCADAS PELA PESCA DE ARRASTO DE BIVALVES NA COSTA ALGARVIA

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Chícharo, L., Alves, F., Regala, J., Chícharo, A. & Gaspar, M. (2002). Alterações nas comunidades bentónicas de substracto móvel provocadas pela pesca de arrasto de bivalves na costa algarvia. *Revista Biol. (Lisboa)* **20**: 37-46.

Estudos relacionados com o impacte ambiental de artes de pesca de arrasto de fundo têm sido efectuados nas últimas décadas e em diversos países. Em Portugal, a pesca de arrasto de bivalves com ganchorra iniciou-se em 1969, desempenhando um papel socio/económico preponderante no Algarve. Com o aumento deste tipo de exploração, também se observa uma crescente preocupação com a gestão de mananciais assim como com o impacte desta actividade sobre o ecossistema marinho. Contudo, pouco ainda se sabe sobre os efeitos do uso destas artes nas comunidades macrobentónicas das águas portuguesas. No presente trabalho abordam-se as alterações a curto e longo prazo nas comunidades macrobentónicas, como consequência da pesca de arrasto de bivalves, compreendendo o efeito a curto prazo na diversidade especifica e respectivo tempo de recuperação, e alterações a longo prazo na estrutura trófica das comunidades. Os resultados apontam para um decréscimo imediato da diversidade especifica e subsequente recuperação detectavel cerca de um mês após a pesca. A estrutura trófica evidencia que áreas intensivamente exploradas são dominadas por comunidades de predadores oportunistas, quando comparada com áreas não exploradas.

Palavras-chave: Impacte de pesca, ganchorra, curto-prazo, longo-prazo, macrobentos.

Chícharo, L., Alves, F., Regala, J., Chícharo, A. & Gaspar, M. (2002). Changes in the soft-bottom benthic communities, due to bivalve dredge fishery off south Portugal. *Revista Biol. (Lisboa)* **20**: 37-46.

Ecosystem impact of shellfish dredge fisheries has been studied worldwide since the last decades and for a number of habitats. In Portugal, this fishery occurs since 1969, and in the Algarve (South Portugal), this fishery plays an important economic-social role. With the increase of the Portuguese clam exploitation, there has been a growing concern about stock management and subsequent impact to the marine ecosystem. However, there is little knowledge on the effects of dredging in the shallow waters of Portugal. In the present study, short-term and long-term changes in community structure due to dredge fishery are approached, encompassing the short-term effects to the macrobenthic diversity, recovery time and long-term effects to trophic structure. Results point to an immediate decrease in diversity and subsequent recovery within the ensuing month. Trophic structure shows that intensely dredged grounds exhibit clear dominance by scavenger communities when compared with unexploited grounds.

Key-words: Impact of fishing, "ganchorra", short-term, long-term, macrobenthic.

INTRODUÇÃO

A pesca de arrasto de fundo afecta as comunidade bentónicas através da remoção e/ou lesão dos organismos que se deparam com a respectiva arte, prejudicando tanto a infauna como o epibentos (BERGMAN & HUP, 1992; TUCK *et al.*, 1998). Por conseguinte, o impacte ambiental da pesca de arrasto tem sido alvo de estudo desde as últimas décadas e em diversos países (CADDY, 1968, 1973; MEYER *et al.*, 1981; KAISER & SPENCER, 1996b; JENNINGS *et al.*, 2001). Como resultado deste esforço em entender os processos subsequentes à pesca de arrasto de fundo, foram detectadas alterações de curto prazo em termos de biodiversidade, composição taxonómica, hábitos alimentares e estrutura comunitária (DE GROOT, 1984; CURRIE & PARRY, 1996; KAISER & SPENCER, 1996b; PRANOVI & GIOVANARDI, 2000). Contudo, as alterações a curto prazo têm sido consideradas particularmente difíceis de avaliar, donde resulta o diminuto número de publicações concernes (TUCK *et al.*, 1998). Não obstante, RODHOUSE *et al.* (1998) afirmaram que as pressões selectivas artificias, resultantes da pesca, promovem um factor de apreciável transformação nas comunidades afectadas.

Em Portugal, a pesca de arrasto de bivalves com ganchorra iniciou-se em 1969, e tem uma importância socio/económica preponderante no Algarve

(GASPAR, 1990, 1996). Actualmente a pesca com ganchorra destina-se à exploração dos bivalves *Chamelea gallina, Donax trunculus, Spisula solida* e *Ensis siliqua* (GASPAR, 1996). Com o aumento deste tipo de exploração, observou-se uma crescente preocupação com a gestão de mananciais assim como com o impacte ambiental desta actividade (GASPAR, 1996). Contudo, pouco ainda se sabe sobre os efeitos resultantes do uso desta arte nas comunidades macrobentónicas das aguas portuguesas.

Estudos recentes têm sugerido um impacto negativo, em termos ecológicos, desta actividade no Algarve (GASPAR *et al.*, 1994, 1998; CHÍCHARO *et al.*, 2001). Contudo, até à data não há dados que permitam quantificar os períodos de recuperação das comunidades macrobentónicas perturbadas, assim como das alterações de grande escala e longo prazo que poderão advir desta pescaria.

Com o presente trabalho pretendem-se estudar as alterações de curto e longo prazo nas comunidades macrobentónicas, como consequência da pesca de arrasto de bivalves, compreendendo o efeito a curto prazo na biodiversidade especifica e respectivo tempo de recuperação, e alterações a longo prazo na estrutura trófica das comunidades.

MÉTODOS

Alterações de curto prazo

Entre Setembro e Novembro de 2000, três áreas replicadas (625 m2 cada) ao largo de Lagos (Fig. 1) foram sujeitas a pesca intensiva durante diversas horas, utilizando os métodos e artes de pesca tradicionais, de acordo com o descrito por Gaspar (1996). Efectuaram-se recolhas aleatórias *In situ* (em mergulho com escafandro autónomo) de 10 quadrats (0.0625 m2 x 0,15 m cada) antes e após a pesca, em cada uma das 3 áreas seleccionadas. Efectuaram-se recolhas posteriores, passadas 24, 48 and 72 horas, 1 e 2 semanas, e 1 e 2 meses nas três áreas perturbadas, e em outras três áreas intocadas (controlo). As recolhas *In situ* foram efectuadas com o auxílio de uma pequena pá, com a qual se transportou o sedimento de uma área e profundidade preestabelecidas, para sacos de malha de 1mm, por forma a efectuar uma crivagem subsequente. As amostras foram posteriormente tríadas e os organismos identificados ao nível específico, em laboratório.

Na análise dos dados, foram usadas quatro medidas univariadas de estrutura comunitária: (1) Abundância de organismos por m2; (2) Número de taxa; (3) Índice de Diversidade de Shannon-Wiener (H') (DAGET, 1974); e (4) Equitabilidade (PIELOU, 1984). Para detectar diferenças significativas ($p<0.05$) entre as amostras recolhidas antes e imediatamente após a pesca, em cada um dos parâmetros, executaram-se 4 ANOVAs de uma via (teste F). Os taxa que mais contribuíram para as dissimilaridades resultantes do impacte da pesca foram determinadas utilizando o método das percentagens de similaridade (SIMPER). No decorrer da análise estatística foi necessário usar os programas STATISTICA 5.0 e PRIMER 5.

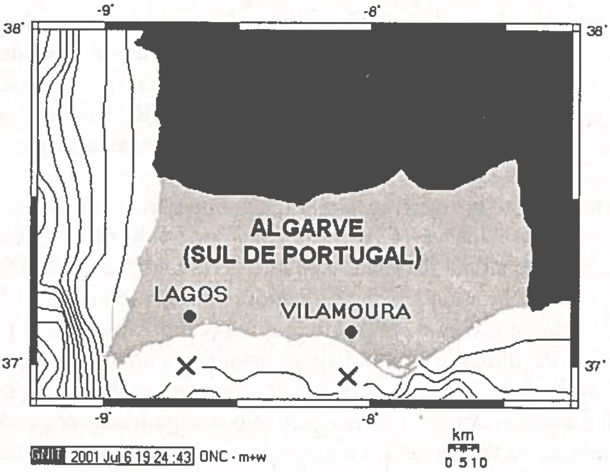

Figura 1 – Localização geográfica das áreas de estudo (X) no Sul de Portugal.

Alterações de longo prazo

Durante o mês de Setembro de 1999 seleccionaram-se duas áreas da costa algarvia sujeitas a diferentes esforços de pesca de arrasto de bivalves com ganchorra: 1) Lagos, onde esta pescaria não ocorre desde 1996; 2) Vilamoura, sujeita a exploração intensa nas últimas décadas (Fig. 1). As áreas foram amostradas, recolhendo aleatoriamente 6 quadrats (0.0625 m2 x0,15 m) em mergulho (SCUBA), por área. Cada replicado foi recolhido, tratado e processado de forma semelhante à descrita acima. Os organismos identificados foram ainda classificados em níveis tróficos de acordo com a bibliografia específica (FEDER, 1981; COMMITO & SHRADER, 1985; RIEDL, 1986; BEMVENUTI, 1994; GAMITO, 1994; MCKILLUP & MCKILLUP, 1994; SPRUNG, 1993, 1994a; FISH & FISH, 1996; SKOELD & ROSENBERG, 1996). As biomassas correspondentes foram calculadas.

A escolha da localização das áreas de amostragem foi efectuada em função dos distintos esforços de pesca e historial pesqueiro, assim como pelas semelhanças hidrológica e sedimentológica, convenientemente salientadas por CHÍCHARO *et al.* (2002). A escolha de zonas relativamente semelhantes em termos físico-químicos e com esforços de pesca contrastantes providencia um potencial comparativo apreciável em termos de ciência experimental (LINDEBOOM & GROOT, 1998).

RESULTADOS

Alterações a curto prazo

Os resultados revelam decréscimo significativo da abundância, nº de taxa e diversidade da macrofauna imediatamente após a acção da ganchorra (Fig. 2 e

Tab. 1). Estas alterações são particularmente significativas tendo em conta os baixos valores de *p* obtidos nas ANOVAs correspondentes. Os dados obtidos também parecem indicar que o dito impacte reflecte-se também na composição específica, tendo em conta que as espécies de macrofauna que mais contribuem para a dissimilaridade (71.17 %) entre as amostras recolhidas antes e após a pesca foram *Siphonoecetes striatus*, *Diogenes pugilator*, *Branchiostoma lanceolatum*, *Urothoe poseidonis* e *Divaricela divaricata*, por ordem de contribuição (Tab. 2).

Tabela 1 – Matriz de resultados respeitantes às quatro ANOVAs de uma via correspondentes às diferenças entre antes e após pesca em termos de Número de indivíduos/m^2, Número de taxa, Índices de Diversidade de Shannon-Wiener (H') e Equitabiladade de distribuição.* Diferenças significativas ($p<0.05$).

ANOVAs de uma via p<0.05	df Effect	MS Effect	df Error	MS Error	F	*p* level
N° ind./m^2	1	126225.07	58	9594.12	13.16	0.001*
N° Taxa	1	50.42	58	5.35	9.42	0.003*
Diversidade (H')	1	0.11	58	0.03	4.03	0.049*
Equitabilidade	1	0.02	58	0.01	3.28	0.075

Tabela 2 – Contribuição das principais espécies para a dissimilaridade entre antes e imediatamente após a pesca com ganchorra (resultados do SIMPER).

SIMPER resultados **TAXA**	Dissimilaridade média = 71.17 **Contrib. %**	**Cum. %**
Siphonoecetes striatus	15.12	15.12
Diogenes pugilator	14.76	29.89
Branchiostoma lanceolatum	10.69	40.58
Urothoe poseidonis	10.33	50.90
Divaricela divaricata	5.53	56.43
Ampelisca brevicornis	4.08	60.51
Echinocardium cordatum	3.82	64.33
Nephtydidae	3.28	67.61
Sipuncula não identificado	2.92	70.53
Chamelea gallina	2.28	72.81

Há indícios de que a estrutura da comunidade macrobentónica é restabelecida num período de 2 meses após a acção da pesca. Ao fim deste período as variações sazonais já eram visíveis, tornando imperceptível o impacte da ganchorra (Fig. 2).

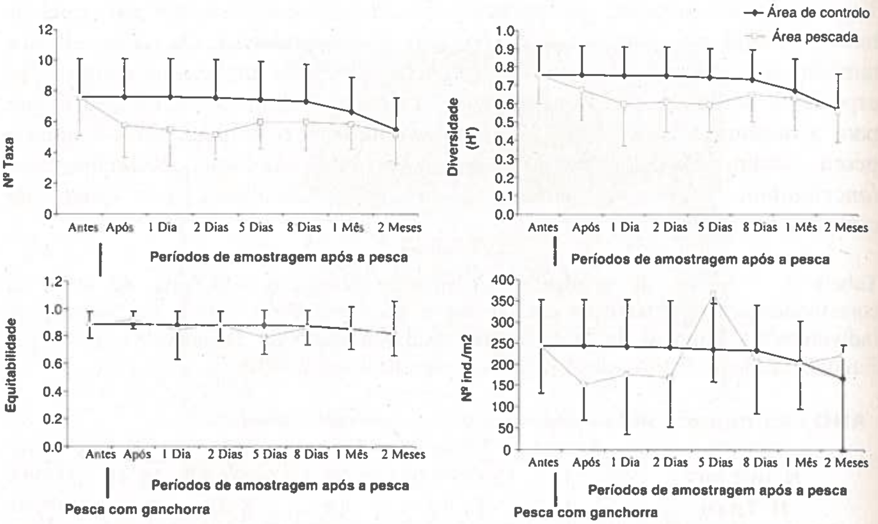

Figura 2 – Alterações e período de recuperação como consequência da pesca com ganchorra.

Alterações a longo prazo

No que respeita aos grupos tróficos, torna-se evidente que os organismos potencialmente "scavengers" e predadores saõ predominantes na zona sujeita ao impacte da pesca, ao passo que os filtradores são particularmente abundantes na zona não impactada. É também de referir que os organismos detritívoros representam uma componente considerável do macrobentos em ambas as zonas amostradas (Fig. 3).

DISCUSSÃO

Alterações a curto prazo

Os resultados obtidos denotam que a pesca de bivalves com ganchorra em Portugal induz alterações substanciais imediatas nas comunidades macrobentónicas de fundo arenoso. Estas observações vêm em concordância com resultados obtidos em outras partes de mundo, observados por BERGMAN & HUP (1992) e PRANOVI & GIOVANARDI (2000), usando metodologias semelhantes.

Durante o período em que as amostragens decorreram, as condições meteorológicas e hidrológicas mantiveram-se apreciavelmente estáveis, para além de as recolhas terem sido efectuadas após a época reprodutiva da maioria das espécies invertebradas amostradas. De acordo com KAISER *et al.* (1998), os efeitos da perturbação induzida pela pesca podem ser mais duradouros dependendo da pesca coincidir com picos de fixação das larvas de invertebrados bentónicos. Não obstante, estes efeitos podem-se revelar negligenciáveis caso ocorram em períodos de intenso hidrodinamismo, tais como tempestades (REES *et al.*, 1977). Esta afirmação vem em conformidade com os resultados que sugerem uma recuperação consideravelmente rápida das comunidades bentónicas, uma vez que se tratam de comunidades sujeitas a perturbações sazonais e esporádicas. Desta forma, e em concordância com diversos autores (LANGTON *et al.*, 1996; KAISER *et al.*, 1998), as implicações destes factores não antropogénicos, deveriam ser integradas nas estratégias de gestão da industria pesqueira e em particular no processo de avaliação de impacte ambiental.

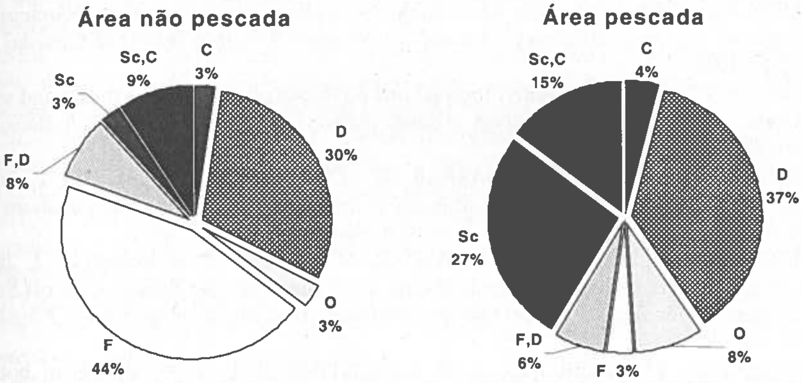

Figura 3 – Percentagem de grupos tróficos detectados nas duas zonas amostradas. D - Detritívoros; S – Filtradores; C – Carnívoros; Sc – "Scavengers"; O – Omnívoros; S,D – Detrívoros e Filtradores; Sc,C –"Scavengers" e Carnívoros.

Alterações a longo prazo

No que respeita às diferenças na estrutura trófica, estas poderão ter origem no facto bem documentado (e.g. KAISER & SPENCER, 1996a; COLLIE *et al.*, 1997; KAISER *et al.*, 1998; LINDEBOOM & GROOT, 1998) de que muitas comunidades oportunistas possam beneficiar da pesca, na medida em que esta poderá providenciar uma fonte alimentar adicional. Desta forma, embora as alterações a curto prazo possam inclusive ter uma duração relativamente curta, existe um outro nível de impacte que necessita ser estudado com mais pormenor. Nomeadamente o das alterações progressivas na estrutura trófica resultantes de um esforço de pesca prolongado e subsequentes consequências para as espécies comerciais.

AGRADECIMENTOS

Ao projecto FAIR ECODREDGE (PL-4465) – "Evaluation and improvement of shellfish dredge design and fishing effort in relation to technical conservation measures and environmental impact", ao abrigo do qual este estudo foi elaborado; ao IPIMAR (Instituto Português de Investigação Marinha) e à Universidade do Algarve por disponibilizarem apoio logístico diverso.

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A COMPARISON STUDY ON THE AGE AND GROWTH OF *ATHERINA BOYERI* RISSO, 1810 AND *ATHERINA PRESBYTER* CUVIER, 1829 IN THE RIA DE AVEIRO, PORTUGAL

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Pombo, L., Lopes, C. & Rebelo, J. E. (2002). A Comparison study on the age and growth of *Atherina boyeri* Risso, 1810 and *Atherina presbyter* Cuvier, 1829 in the Ria de Aveiro, Portugal. *Revista Biol. (Lisboa)* **20**: 47-58.

Atherina boyeri and *Atherina presbyter* show similar morphology and anatomy, which difficult the taxonomic species identification by conventional tools. The ecological behaviour and biological features are markedly distinct in both species. In the Ria de Aveiro, a typical coastal estuarine lagoon, *A. boyeri* and *A. presbyter* are comprised in the four most abundant species. Ecology, age and growth of both species were compared based on sampling carried out during 13th months (November 1998/99) at nine stations covering all the lagoon area. The frequency of abundance per length, and the fish and otolith length-weight relationships were significantly different in both species. *A. boyeri* reached 11.6 cm in 3 years while *A. presbyter* reached 15.1 cm in 4 years. For the same length, *A. boyeri* (b= 3.4) was heavier than *A. presbyter* (b= 3.1). The von Bertalanffy growth parameters for *A. boyeri* were L_∞= 11.55 cm, k= 0.099 year^{-1}, and t_0= -3.797 years and for *A. presbyter* were L_∞= 15.75 cm, k= 0.138 year^{-1}, and t_0= -2.501 years. *A. boyeri* and *A. presbyter* grew faster during the first year of life (66% and 43%, respectively). The observed and theoretical length and age were quite similar to both species and identical to different Atlantic and Mediterranean systems.

Key words: ecology, age, growth, *Atherinidae*, Ria de Aveiro.

Pombo, L., Lopes, C. & Rebelo, J. E. (2002). Estudo comparativo de idade e crescimento de *Atherina boyeri* Risso, 1810 e *Atherina presbyter* Cuvier, 1829 na Ria de Aveiro, Portugal. *Revista Biol. (Lisboa)* **20**: 47-58.

Atherina boyeri e *Atherina presbyter* apresentam morfologia e anatomia semelhantes, o que dificulta a identificação das espécies a partir das chaves taxonómicas convencionais. O comportamento ecológico e as características biológicas são nitidamente distintas em ambas as espécies. Na Ria de Aveiro, um ambiente estuarino do tipo lagunar, *Atherina boyeri* e *Atherina presbyter* incluem-se nas quatro espécies mais abundantes. Foi realizado um estudo comparativo da ecologia, idade e crescimento de ambas as espécies, tendo sido capturados exemplares durante 13 meses (Novembro de 1998/99), em nove estações de amostragem que cobrem toda a área lagunar. A frequência de abundância por comprimento, e as relações comprimento-peso dos peixes e dos otólitos foram significativamente diferentes em ambas espécies. *Atherina boyeri* atingiu 11,6 cm em 3 anos enquanto que *Atherina presbyter* atingiu 15,1 cm em 4 anos. Para o mesmo comprimento, *Atherina boyeri* (b= 3,4) foi mais pesada do que a *Atherina presbyter* (b= 3,1). Os parâmetros de crescimento de von Bertalanffy para *Atherina boyeri* foram L_{∞}= 11,55 cm, k= 0,099 ano^{-1}, e t_0= -3,797 anos e para *Atherina presbyter* foram L_{∞}= 15,75 cm, k= 0,138 ano^{-1}, e t_0= -2,501 anos. *Atherina boyeri* e *Atherina presbyter* cresceram mais rapidamente durante o primeiro ano de vida (66% e 43%, respectivamente). Os comprimentos teóricos e observados e as idades foram muito semelhantes em ambas as espécies e muito aproximados aos dos diferentes sistemas Atlântico e Mediterrâneo.

Palavras-chave: ecologia, idade, crescimento, *Atherinidae*, Ria de Aveiro.

INTRODUCTION

The Atherinid fish species, big-scale sand smelt *Atherina boyeri* Risso, 1810 and sand smelt *Atherina presbyter* Cuvier, 1829, exhibit a similar morphology and anatomy. The taxonomic identification by the common morphometric characteristics, as head length, number of scales in longitudinal series, and number of vertebrae (WHITEHEAD *et al.*, 1986; BAUCHOT & PRAS, 1987), is ambiguous and leads some doubts. BAMBER & HENDERSON (1985), inclusively, proposed the synonymy of *A. boyeri* with *A. presbyter*. However, recent studies approaching genetic and biological aspects have demonstrated differences between these two populations, consistent with the existence of two species (CREECH, 1991, 1992). The species comprise distinct ecology behaviour and different geographical distribution. *A. boyeri* was described as a truly resident species, preferring littoral, estuaries, lagoons, and inshore areas forming land-locked populations where it spends its entire life (ELLIOTT & DEWAILLY, 1995, WHITEHEAD *et al.*, 1986). The distribution area includes the Mediterranean and adjacent seas, the Black Sea and the Atlantic, from south of Spain to Morocco and Madeira. Isolated populations can be found on coasts of England and Netherlands (WHITEHEAD *et al.*, 1986; ROSECCHI & CRIVELLI, 1992, 1995; LEONARDOS & SINIS, 2000; LEONARDOS, 2001). *A. presbyter* was described as a marine juvenile migrant species, which uses estuaries and coastal lagoons primarily as a nursery ground. Much of its adult life is spending at sea, but often returning seasonally to the estuary (ELLIOTT & DEWAILLY, 1995). It occurs along the European and African coasts of the Atlantic Ocean, from the Kattegat (rare) and Scotland to Morocco, and around Madeira, Canaries and Cape Verde Islands. It is rare in the Mediterranean, found off Gibraltar, France and Tunisia (WHITEHEAD *et al.*, 1986; LORENZO & PAJUELO, 1999; PAJUELO & LORENZO, 2000).

To the Portuguese coast, besides the Ria de Aveiro, only *A. presbyter* was referred, namely in Tejo estuary (COSTA, 1986; COSTA & BRUXELAS, 1989), in Sado estuary (CABRAL, 1999) and in Ria Formosa (MONTEIRO *et al.*, 1987; SANTOS *et al.*, 1996). In the Ria de Aveiro, *A. boyeri* was turned visible since 1992 (REBELO, 1992; POMBO & REBELO, 2000; CRUZ & REBELO, 2000), so before that only *A. presbyter* was described to the lagoon (NOBRE *et al.*, 1915; OSÓRIO, 1912; ARRUDA *et al.*, 1988). Nowadays, both species are comprised in the four most abundant species of the lagoon.

This paper intends to distinguish two Atherinids populations (*A. boyeri* and *A. presbyter*) through their ecology, age and growth patterns, according to their occurrence in a typical coastal estuarine lagoon.

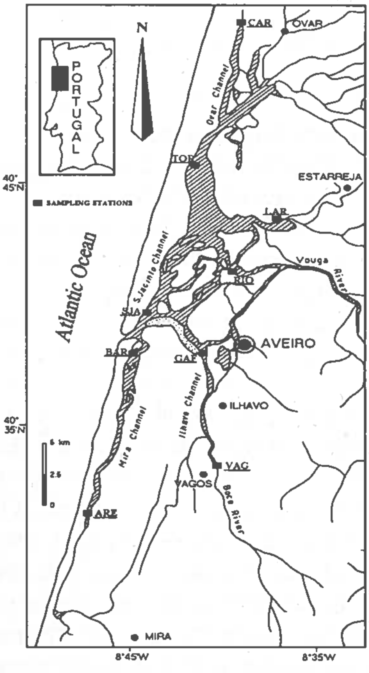

Figure 1. Map of the Ria de Aveiro lagoon showing sampling stations.

Study area

The Ria de Aveiro (fig. 1), on the west coast of Portugal is a coastal estuarine lagoon. A maximum length of 45 km and maximum width of 11 km delimit a total area between 43 km² (low tide) and 47 km² (high tide). The water volume is 70 million m^3, with an oceanic tidal contribution of between 25 million m^3 (neap-tides) to 90 million m^3 (higher spring tides). Four rivers (Vouga, Antuã, Caster and Boco) and uncounted streams flow between 3 m^3 s^{-1} and 60 m^3 s^{-1}, due to the influence of seasonal precipitation and run-off patterns (BARROSA, 1980). The depth varies between 0.6 m and 10.0 m. The mean substratum is silt, however sandy close to the lagoon's mouth and with immerses vegetation in the north extreme.

The industrial and fishing ports, and the industries and population in the watershed are sources of three main pollution types: organic and chemical pollution, chemical pollution, and microbial contaminants (BARROSA, 1980; LUCAS *et al.*, 1986; BORREGO *et al.*, 1994). The Ria de Aveiro is economically important because of its fisheries, industry, agriculture, sea farming, tourism and, more recently, aquaculture. With a good communication with the sea that guarantees seasonally the fish recruitment, the lagoon is an area of considerable fish exploitation (commercial and recreational).

MATERIAL AND METHODS

Fish were collected monthly, from November 1998 to November 1999, at nine selected stations (fig. 1) covering all the lagoon area: near the mouth of the lagoon (BAR, GAF and SJA); at the edges of the main channels (ARE, CAR and VAG); in the main freshwater area, highly organically enriched (RIO); in the region of higher levels of industrial pollution (LAR), and approximately in the middle of the longest channel (TOR).

Samples were taken in triplicate at low spring tides with a traditionally beach-seine type net used in the region, “chincha”.

Fish were preserved freezing. At the laboratory, specimens were identified according to the taxonomic keys of WHITEHEAD *et al.* (1986) and BAUCHOT & PRAS (1987), measured to the nearest 0.01 cm (total length, *L*), and weighed to the nearest 0.1 g (total weight, *W*).

Otoliths and scales themselves are largely used in age and growth studies. Alternating opaque and translucent rings are formed during the summer and winter, respectively (CASSELMAN, 1983), with every pair of these rings indicating approximately one year's growth. The otoliths and scales considered for age studies were removed from fish obtained in aleatory sub-samples of 3 specimens of each species, from each station and month sampling, of the same length class. Otoliths were stored dry in properly labelled envelopes. Scales were removed from the base of the pectoral fin in the longitudinal line. Otoliths were measured, from the anterior tip to the posterior projection, to the nearest 0.01 mm, and weighed, to the nearest 0.1 mg. Right otoliths were examined, except in abnormal cases (*i.e.* broken), where the left was examined, under reflected light with a stereo microscope (*OLYMPUS SZ60-PT*) having a micrometer eyepiece. The number of opaque zones (winter rings, appearing bright under reflected light) and the presence of a marginal translucent zone (summer rings, appearing dark under reflected light) were registered. Otoliths were compared to scale ageing readings and only coincided readings were considered, although the percentage of coincidence has not been recorded.

The length distributions of both species were compared by Kolmogorov-Smirnov (K-S) two-sample test (SOKAL & ROHLF, 1995; ZAR, 2000).

The fish length-weight relationship was calculated using all individuals sampled, following the Gulland's equation (GULLAND, 1969): $W = a\ L^{b}$, where W is the total body weight (g), L is the total length (cm), and a and b are the regression coefficients. The same equation was adopted for otoliths length-weight relationship to analyse if the otoliths growth were similar to the fish growth. The comparison of slopes of fish length-weight and the otolith length-weight relationships was carried out by using the Student's t test with log-transforming data: $t = \frac{b_1 - b_2}{S_{b_1 - b_2}}$, where b_1 and b_2 are the regression coefficients and S_{b1-b2} is the standard error of the difference between regression coefficients (ZAR, 2000).

One-way ANOVA was carried out, for each species, to test the significance of difference between fish length age groups. When the normality test failed a Kruskal-Wallis One Way Analysis of Variance on Ranks was used (SOKAL & ROHLF, 1995; ZAR, 2000). The package *Sigma Stat software* (version 2.03) (*ANON.*, 1997) was used for these analyses.

Age growth was expressed in terms of the von Bertalanffy equation (von Bertalanffy, 1938): $L_t = L_\infty \left[1 - e^{-k(t-t_0)}\right]$, where L_t is the fish length at age t, L_∞ is the maximum asymptotic length, k is the growth curvature parameter, and t_0 is the computed age where the fish has no growth both in length and weight. The growth parameters were estimated considering all the individuals with age recorded. The package used for these analyses was ELEFAN I (*Electronic Length Frequency Analysis*), included in FISAT, developed by Pauly & David (PAULY, 1987).

RESULTS

A total of 2503 specimens of *A. boyeri* and 1169 specimens of *A. presbyter* were caught and considered for the fish analyses (length frequency distribution and length-weight relationship). From all the caught fish, a sub-sample of 350 and 477 otoliths were removed and examined, for *A. boyeri* and for *A. presbyter*, respectively.

Length frequency distribution

The length composition (fig. 2) significantly varied between the two species (K-S test: d max = 0.4, $p<0.001$). In *A. boyeri* the minimum and maximum lengths were 1.56 cm and 11.62 cm, respectively, while in *A. presbyter* were 4.07 cm and 15.10 cm, respectively. The most abundant length classes were 6.5-8.5 cm, in *A. boyeri*, and 10.5-12.5 cm, in *A. presbyter*.

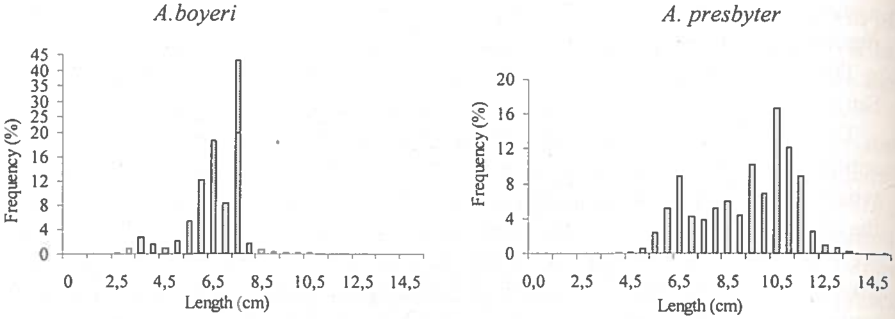

Figure 2. Relative frequency of abundance per length in *A. boyeri* (N= 2503) and *A. presbyter* (N=1169).

Length-weight relationship

Fish length-weight relationship, an expression of growth, was established separately to the two species. The length-weight equation, for *A. boyeri*, was $W= 3.3x10^{-3}.L^{3.35}$ ($r^2= 0.967$) and for *A. presbyter* was $W= 5.5x10^{-3}.L^{3.09}$ ($r^2= 0.973$) (table I). Both equations showed that the weight grew allometrically with the length; the *b* values ($b>3$) showed that species grew faster in weight than in length (TESCH, 1971). For the same length, *A. boyeri* ($b= 3.4$) became heavier than *A. presbyter* ($b= 3.1$). The comparative analysis of the regressions revealed statistically different slopes ($t= 28.02$, $p<0.001$, df=3670).

Table I. Number of species sampled (N) and length-weight relationship parameters (*a* and *b*) for *A. boyeri* and *A. presbyter* in Atlantic and Mediterranean systems (adapted from Leonardos & Sinis, 2000)

Authors	Study area	*N*	*a*	*b*
	Atherina boyeri			
Fernandez-Delgado *et al*., 1988	Guadalquivir River, Spain	2510	$7.0x\ 10^{-3}$	2.98
Creech, 1992	Aberthaw Lagoon S. Wales	329	$3.5x10^{-6}$	3.27
Leonardos *et al*., 1993	Trichonis Lake W. Greece	572	$3.8x10^{-6}$	3.45
Leonardos & Sinis, 2000	Mesolongi, Etolikon W. Greece	4269	$4.2x10^{-3}$	3.15
Present study	Ria de Aveiro, Portugal	2503	$3.3x10^{-3}$	3.35
	Atherina presbyter			
Pajuelo & Lorenzo, 2000	Canary Islands	1028	$6.1x10^{-3}$	3.08
Present study	Ria de Aveiro, Portugal	1169	$5.5x10^{-3}$	3.09

Otoliths length-weight relationship

The Gulland's growth equation to fish length and weight was fitted to the otoliths growth. The otoliths length-weight equation, for *A. boyeri*, was $OW= 1.7x10^{-2}.OL^{2.9}$ ($r^2= 0.92$) and for *A. presbyter* was $OW= 1.3x10^{-2}.L^{3.1}$ ($r^2= 0.96$).

A. boyeri's otoliths grew faster in length than in weight (b= 2.9). The opposite occurred to *A. presbyter* (b= 3.1). The comparative analysis of the otolith regressions revealed statistically different slopes (*t*= 6.15, p<0.001, df= 825).

The otolith growth followed the fish growth. The smallest fish (3.39 cm in *A. boyeri* and 4.59 cm in *A. presbyter*) corresponded to the smallest otolith length (1.28 and 1.83 mm respectively), and the longest fish (11.62 cm in *A. boyeri* and 15.10 cm in *A. presbyter*) corresponded to the longest otolith length (4.25 and 5.40 mm respectively).

The relationship between fish length and otolith length is similar in both species (*t*= -3.760, p>0.05), *i.e.*, the otolith length attends the fish length in the same proportions in both species.

Age and growth

Intending to compare the length-age growth, the fish lengths were divided into one-year age groups. In both species the fish lengths were statistically different between age groups. For *A. boyeri* (Kruskal-Wallis One Way Analysis of Variance on Ranks was used as normality test failed for ANOVA) H=156.66, p<0.001, with 3 degrees of freedom, and for *A. presbyter*, F= 416.90, p<0.001, df= 475.

The mean incremental growth per age group was higher in *A. boyeri*, between 0.63 cm and 1.44 cm, than in *A. presbyter*, between 1.77 cm and 2.60 cm (fig. 3). The proportion of growth was calculated in each age group. The first year of life growth in *A. boyeri* and *A. presbyter* (66% and 43%, respectively) (table III) was remarkably faster than the consequent age groups.

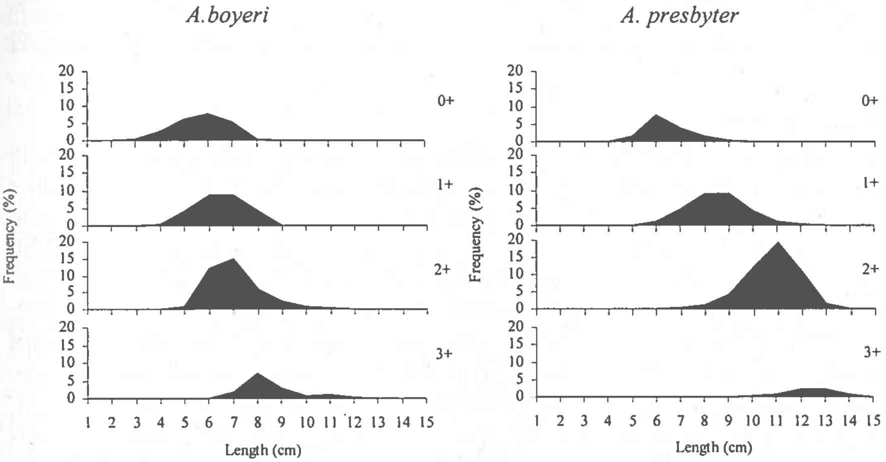

Figure 3. Relative frequency of abundance per length and age in *A. boyeri* (N= 350) and *Atherina presbyter* (N= 477).

In spite of the non-selectivity of the used methodology, in both species, the 2^+ age group was found to be the most abundant, followed by 1^+, 0^+ and 3^+ age groups. The 4^+ age group was just found in *A. presbyter*, but only in a unique specimen, though it is not represented in figure 3.

The von Bertalanffy growth equation was established for *A. boyeri* and *A. presbyter* (table II). In both species, the theoretical lengths for *A. boyeri*, 11.55 cm, and for *A. presbyter*, 15.75 cm, were realistic and agreed to the largest specimen caught during this study, 11.62 cm and 15.10 cm, respectively. Theoretical length values were compared with the fish length collected from nature (table III) and found to coincide.

Table II – Maximum age and von Bertalanffy parameters ($L\infty$, K, and t_0) for *A. boyeri* and *A. presbyter* in Atlantic and Mediterranean systems (adapted from Leonardos & Sinis, 2000)

Authors	Study area	Maximum age (years)	$L\infty$ (cm)	K (year^{-1})	t_0 (years)
	Atherina boyeri				
Fernandez-Delgado *et al*., 1988	Guadalquivir River, Spain	2	-	-	-
Creech, 1992	Aberthaw Lagoon S. Wales	2	9.2	-	-
Leonardos *et al*., 1993	Trichonis Lake W. Greece	4	12.32	0.374	0.018
Leonardos & Sinis, 2000	Mesolongi, Etolikon W. Greece	3	11.58	0.24	-1.27
Present study	Ria de Aveiro, Portugal	3	11.55	0.099	-3.797
	Atherina presbyter				
Pajuelo & Lorenzo, 2000	Canary Islands	3	12.18	0.79	0.21
Present study	Ria de Aveiro, Portugal	4	15.75	0.138	-2.501

DISCUSSION

Biological features, as length, weight, growth and age are distinctive in both species. The length composition was markedly different. *A. boyeri* was smaller but heavier than *A. presbyter*. The growth in length was generally slow and lower in *A. boyeri* than in *A. presbyter*, except during the juvenile period. Atherinids, in general, experienced a fast early growth rate through the first year of age (FERNANDEZ-DELGADO *et al.*, 1988; CREECH, 1992; LEONARDOS & SINIS, 2000), after which the annual growth rate drops rapidly, resulting of less energy available for somatic growth associated to sexual maturity (PAJUELO & LORENZO, 2000). In the first year of life, *A. boyeri* reached approximately $^2/_3$ of the maximum length. The same result was obtained to Greek Mediterranean lagoons (Mesolongi and Etolikon) (LEONARDOS & SINIS, 2000). *A. presbyter* reached less than half of maximum length in the first year of life, opposing to 60% registered in the Canary Islands (LORENZO & PAGUELO, 1999).

Table III. Comparison of observed caught in nature mean length values and theoretical calculated length values according to age groups. The percentage of growth was calculated with the observed values.

Age groups	N	Observed	Theoretical	% Growth
A. boyeri				
0^+	79	5.6	5.45	65.49
1^+	90	6.48	6.58	10.29
2^+	131	7.11	7.6	7.37
3^+	50	8.55	8.52	16.84
4^+	-	-	-	-
A. presbyter				
0^+	71	6.51	6.61	43.11
1^+	140	8.47	8.68	12.98
2^+	238	10.73	10.48	14.97
3^+	27	12.5	12.05	11.72
4^+	1	15.1	13.41	17.22

In *A. boyeri* the *b* value was very close to the established in the Atlantic Aberthaw lagoon (CREECH, 1992), in the Mediterranean Mesolongi and Etolikon lagoons (LEONARDOS & SINIS, 2000) and in the Mediterranean lake Trichonis (LEONARDOS *et al.*, 1993) (table I). The *A. presbyter b* value was very similar to the Atlantic Canary Islands (PAJUELO & LORENZO, 2000), in the south limit of the species distribution (table I).

The biometric otolith analysis showed that the otoliths growth followed the fish growth independently of the species or the maximum length that each species can reach. However, as well as for the fish length/weight, the significantly difference between *A. boyeri* and *A. presbyter* otolith length/weight relationship contribute to distinguish the two species.

The age and length composition of both species was quite different. The estimated L_∞ (11.55 cm for *A. boyeri* and 15.75 cm for *A. presbyter*) was in close agreement with the maximum length recorded (L_{max}= 11.6 cm and L_{max}= 15.10 cm, respectively). The theoretical maximum length values were reasonably higher than the maximum sampled size. In *A. boyeri*, L_∞ was very similar to the Greek Mediterranean lagoons (Mesolongi and Etolikon lagoons (LEONARDOS & SINIS, 2000)) and deeply higher than in Wales (Aberthaw lagoon (CREECH, 1992)) (table II). In contrast, the Mediterranean lake system (Trichonis lake (LEONARDOS *et al.*, 1993)) exhibited a maximum value of L_∞. In *A. presbyter*,

L_∞ was higher than in the Atlantic Canary Islands (PAJUELO & LORENZO, 2000) (table II).

It may be concluded that *A. boyeri* and *A. presbyter* are sympatric species that might occur at the same space and time. Both species show similar macroscopic shape, leading to an eventual generalised confusion according to conventional species taxonomic identification. They exhibit, however, distinct ecological behaviour and biological characteristics. The abundance, the fish and otolith length-weight relationships and the age and growth parameters are markedly distinct in both species. Some of these ecological and biological parameters can be generally useful as accessory tools in the taxonomic identification of the Atherinids species.

ACKNOWLEDGEMENTS

We would like to thank the *University of Aveiro* and the *Foundation for Science and Technology* for their financial support of this research. The authors are also very grateful to Rui Cruz, Marina Dolbeth and Florent Garnerot for their field and lab collaboration.

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DIET COMPARISON BETWEEN *ATHERINA BOYERI* RISSO, 1810 AND *ATHERINA PRESBYTER* CUVIER, 1829 IN THE RIA DE AVEIRO, PORTUGAL

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Pombo, L., Dias, C. & Rebelo, J. E. (2002). Diet comparison between *Atherina boyeri* Risso, 1810 and *Atherina presbyter* Cuvier, 1829 in the Ria de Aveiro, Portugal. *Revista Biol. (Lisboa)* **20**: 59-72.

This work compares the diet of *A. boyeri and A. presbyter, of family* Atherinidae, one of the most representative fish families present in the lagoon. Samples were taken at nine stations, covering all the lagoon area, during 13th months (November 1998/99). A total of 346 stomachs of *A. boyeri* and 493 stomachs of *A. presbyter* were analysed. The modified Costello method was used to characterise the diets of the two species. The diet composition of both species was similar and was composed by a large variety of food items. The main food item was crustaceans, representing almost 55% and 47% of total ingested food for *A. boyeri* and *A. presbyter*, respectively. Within crustaceans, small benthonic crustaceans were the most representative group. The values obtained in the Levins measure were wide, (7.24, for *A. boyeri* and 8.99, for *A. presbyter*), reflecting the generalised feeding strategy in both species. In spite of both Atherinids species consume the same food items, those items did not represent the same importance for both species, such as annelids, macrocrustaceans, detritus and eggs. Some prey items were representative of particular age groups for both species (Cumacea for 0^+-group, Decapoda for 2^+ and 3^+-group, and *Praunus flexuosus* for 2^+-group). Comparing the two species, 7 food items showed different representativeness for a particular age group (gastropods, larvae of insects, small pelagic crustaceans, macrocrustaceans, fishes, detritus, and eggs). The distinct importance of each food item for the diet of *A. boyeri* and *A. presbyter* improves the idea that they do not compete for the existing food, which certainly is largely available in the environment. Thus, it is possible to induce that the non-competition between these two sympatric species leads to a possible co-habitation in the same ecosystem, at the same time and space, as it occurs in the studied estuarine coastal lagoon, Ria de Aveiro.

Key words: diet analysis, *Atherina boyeri*, *Atherina presbyter*, Atherinidae, Ria de Aveiro, Portugal.

Pombo, L., Dias, C. & Rebelo, J. E. (2002). Comparação da dieta entre *Atherina boyeri* Risso, 1810 e *Atherina presbyter* Cuvier, 1829 na Ria de Aveiro, Portugal. *Revista Biol. (Lisboa)* **20**: 59-72.

Este trabalho compara a dieta de *A. boyeri* e *A. presbyter* da família Atherinidae, uma das famílias de peixes mais representativas da laguna. As amostragens foram realizadas em nove estações de amostragem que abrangem toda a área lagunar, durante treze meses (Novembro 1998/99). Foram analisados um total de 346 estômagos de *A. boyeri* e 493 estômagos de *A. presbyter*. Foi utilizado o método modificado de Costello para caracterizar a dieta de ambas as espécies. A sua composição da dieta foi semelhante e foi composta por uma vasta variedade de itens alimentares. Os crustáceos constituíram o item alimentar principal, representando cerca de 55% e 47% do alimento total ingerido para *A. boyeri* e *A. presbyter*, respectivamente. Dentro do grupo dos crustáceos, os pequenos crustáceos bentónicos foram o grupo mais representativo. Os valores alargados do índice de Levins (7,24 para *A. boyeri* e 8,99 para *A. presbyter*) reflectiram uma estratégia alimentar generalizada em ambas as espécies. Apesar de consumirem os mesmos itens alimentares, esses itens não representaram a mesma importância para as duas espécies, tais como os anelídeos, os macrocrustáceos, os detritos e os ovos. Certos itens de presas foram representativos de um determinado grupo de idade para as duas espécies (Cumacea para o grupo 0^+, Decapoda para os grupos 2^+ e 3^+, e *Praunus flexuosus* para o grupo 2^+). Comparando as duas espécies, 7 itens alimentares evidenciaram uma diferente representatividade para um determinado grupo de idade, (gastrópodes, larvas de insectos, pequenos crustáceos pelágicos, macrocrustáceos, peixes, detritos e ovos). O facto de cada item alimentar assumir importância distinta para cada uma das espécies, reforça a ideia que elas não competem pelo alimento existente, que provavelmente está disponível em grandes quantidades. Por isso, é legítimo afirmar que a não competição entre estas espécies simpátricas faz com que possam co-habitar no mesmo ecossistema, no mesmo tempo e espaço, tal como acontece na laguna costeira estuarina, Ria de Aveiro.

Palavras-chave: análise da dieta, *Atherina boyeri*, *Atherina presbyter*, Atherinidae, Ria de Aveiro, Portugal.

INTRODUCTION

The big-scale sand smelt, *Atherina boyeri* Risso, 1810, and the sand smelt, *Atherina presbyter* Cuvier, 1829, exhibit very similar morphology and anatomy. The identification by the number of scales in longitudinal series and number of vertebrae (QUIGNARD & PRAS, 1986 Atherinidae. Pages. 1207-1210, *In* WHITEHEAD *et al*., 1986 & BAUCHOT & PRAS, 1987) is ambiguous and leads to some doubts. BAMBER & HENDERSON (1985), inclusively, proposed the synonymy of *A. presbyter* with *A. boyeri*. However, recent studies concerned with age and growth, and other genetical and ecological aspects have demonstrated differences between populations, consistent with the existence of two species (CREECH, 1991, 1992).

The information about Atherinids is scarce; the knowledge of food diet of fish from its natural environment is essential for the understanding of its range in growth, some aspects of reproduction, and the whole behaviour, namely migrations (KARA & DERBAL, 1996).

According to NIKOLSKY (1963), the fish can be classified as euryphagic, stenophagic, or monophagic, depending on the variety of the food consumed. Furthermore, AZEVEDO (1995) classifies them as herbivorous, macro or micro-omnivorous, or macro, meso or micro-carnivorous, according to the type of prey consumed.

This work intends to analyse the biotic relationship (competition, trophodynamics and food web position) of the studied species, in order to emphasise the distinction of these sympatric species.

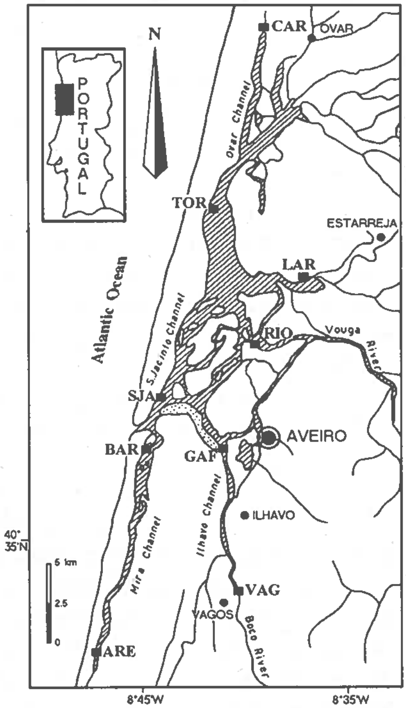

Figure 1. Map of the Ria de Aveiro lagoon, showing sampling stations.

Study area

The Ria de Aveiro (fig.1), on the west coast of Portugal is a coastal estuarine lagoon with a maximum length of 45 km, maximum width of 11 km, and covering a total area between 43 km² (low tide) and 47 km² (high tide). The water volume is 70 million m^3, with an oceanic tidal contribution between 25 million m^3 (neap-tides) and 90 million m^3 (higher spring tides). Four rivers (Vouga, Antuã, Caster and Boco) and uncounted streams flow into it at a rate between 3 $m^3 s^{-1}$ and 60 $m^3 s^{-1}$, depending on the seasonal precipitation and run-off patterns (BARROSA, 1980).

MATERIAL AND METHODS

The samples were monthly caught, from November 1998 to November 1999, at nine sampling stations (fig.1). Samples were taken in triplicate, at low tide, with "chincha", a traditional beach-seine net. The fishes were preserved by freezing. In the laboratory, each specimen was identified according to QUIGNARD & PRAS (1986) Atherinidae. Pages. 1207-1210, *In* WHITEHEAD *et al.* (1986); and otoliths and stomachs were removed.

Otoliths were examined with a stereomicroscope (*Olympus SZ60-PT*) (630x) and the age was estimated through the counting of translucent zones (summer rings, with a dark appearance under reflected light).

The stomachs were weighted with an electronic scale (*And-HR-60*) to the nearest 0.0001 g, before and after extraction of the stomach content, which was preserved in a 70% ethanol solution. The prey items were identified to the lowest possible taxonomic level, according to CAMBELL, 1988 and HAYWARD & RYLAND, 1998, with a stereo microscope (*Olympus SZ60-PT*) (630x) and an optic microscope (*Olympus CH 30*) (1000x). In spite of prey have been identified to the lowest taxonomic level, they were grouped into 13 categories in order to become more clear the interpretation of data. Thus, it was considered annelids, gastropods, bivalves, insects, crustaceans, fishes, plants, detritus and eggs. As crustaceans constitute a large group, 4 sub-groups were considered: the microcrustaceans (Cumacea, Copepoda, Ostracoda and larvae), small benthonic crustaceans (Isopoda and Amphipoda), small pelagic crustaceans (Mysidacea) and macrocrustaceans (Decapoda) (table I).

The level of identification of food items is a function of the digestion stage of prey (ROSECCHI & NOUAZE, 1985). To evaluate the amount of each prey, some rules were considered: i) some items, like algae, parts of fish and unidentifiable material, were considered as a single unit (ARIAS, 1980); ii) some smooth invertebrates, annelids and polychaetes, as they are rapidly digested, appeared in fragments or as a semi-digested mass; but they were considered as a single unit as well (KTARI *et al.*, 1978); iii) the number of some crustaceans, such as mysids, was determined by dividing the number of dispersed eyes per two (KARA & DERBAL, 1996); iv) other crustaceans were identified and counted from some parts of the body that allow their individualization, as the heads for amphipods, or cephalothoraxes or abdominal for mysids and decapods (KARA & DERBAL, 1996); v) a small number of

Table I. List of identified prey grouped taxonomicaly in *A. boyeri* and *A. presbyter* with number of individual prey items (Ni), percentage of frequency of occurrence (FO), percentage of prey-specific abundance (Pi), total stomachs analysed with identified prey, and empty stomachs.

Species	*A. boyeri*			*A. presbyter*		
Prey	Ni	FO (%)	Pi (%)	Ni	FO (%)	Pi (%)
Phylum Annelida	147	31.5	30.4	462	38.3	50.0
Class Polychaeta	107	23.1	30.5	368	32.1	52.5
Phyllum Mollusca						
Class Gastropoda	90	4.6	30.3	138	7.3	37.8
Class Bivalvia	329	9.7	80.0	563	20.3	53.9
Phyllum Artropoda						
Class Insecta	52	11.3	29.0	255	16.6	34.1
Family Formicidae	–	–	–	26	0.8	53.1
Insecta larva	35	10.0	10.2	55	7.6	22.6
Phyllum Crustacea						
Microcrustacea						
Order Cumacea	6	0.4	31.6	3	0.6	7.5
Class Ostracoda	10	1.7	17.5	18	1.7	35.3
Class Copepoda	350	8.0	76.3	226	4.5	60.9
Crustacea larvae	–	–	–	1	0.3	33.3
Small benthonic crustacea						
Order Isopoda	1	0.4	11.1	6	0.8	31.6
Gnatia sp.	–	–	–	8	0.6	28.6
Idotea sp.	7	1.3	46.7	6	1.4	6.4
Sphaeroma sp.	420	31.9	48.4	459	22.0	53.7
Order Amphipoda	14	2.5	56.0	102	0.3	31.8
Ampithoe sp.	–	–	–	1	0.3	20.0
Corophium sp.	–	–	–	1	0.3	100.0
Gammarus sp.	–	–	–	6	0.6	35.3
Small pelagic crustacea						
Order Mysidacea	25	6.7	25.8	58	8.7	15.4
Gastrosaccus sp.	–	–	–	6	0.3	100.0
Neomysis sp.	–	–	–	1	0.3	20.0
Praunus flexuosus	5	0.8	38.5	4	0.6	44.4
Macrocrustacea						
Order Decapoda	–	–	–	12	2.3	30.8
Crangon crangon	7	0.8	77.8	6	1.1	37.5
Fishes	21	6.7	34.4	22	5.6	32.4
Plants	21	8.4	11.1	26	7.0	25.0
Detritus	15	4.2	55.6	26	4.8	76.5
Eggs	43	5.5	18.5	130	5.6	52.8
Total stomachs analysed		346			493	
Empty stomachs		1			2	
Total stomachs with identified prey		238			355	

other items found in stomachs (e.g. molluscs shells and both larval and post larval crustaceans, and parasitic worms) were considered as miscellaneous or non-food items, probably ingested incidentally in normal feeding; vi) prey too digested for unequivocal identification were assigned to one of the appropriate main prey categories (fish, crustaceans, molluscs, annelids and insects) as "remains" and included in the quantification of the contributions of those main categories to the diet (FRANKS *et al.*, 1996).

The feeding strategy was analysed using the modified Costello method (AMUNDSEN *et al.*, 1996) (fig. 2), which consists in a bidimensional graphical representation of the prey abundance and their frequency of occurrence, both in percentual terms. The same method was used to analyse the age diet consumption.

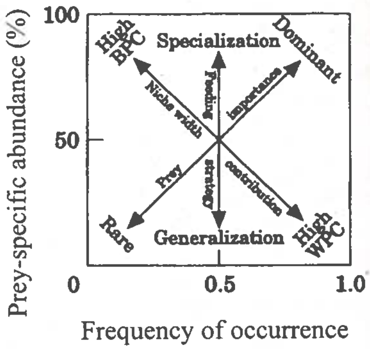

Figure 2. Diagram for the interpretation of feeding strategies, by the modified Costello method (AMUNDSEN *et. al.*, 1996).

The breadth niche was estimated using the Levins index (Levins *in* Krebs, 1989): $B = \dfrac{Y^2}{\sum N_j^2}$, where B is the Levins' measure of niche breadth; N_j represents the number of individuals found in or using resource state j and $Y = \sum N_j$, which indicates the total number of individuals sampled. After measuring the Levins index, it was preceded to the standardization in order to express the niche breadth on a scale from 0 to 1.0. The formula used was $B_A = \dfrac{B-1}{n-1}$, where B_A is the Levins' standardized niche breadth; B is the Levins' measure of niche breadth and n represents the number of possible resources states (Krebs, 1989).

RESULTS

Feeding strategy

A total of 346 stomachs of *A. boyeri* and 493 stomachs of *A. presbyter* were analysed, from which only 1 and 2 were empty respectively in *A. boyeri* and *A. presbyter*. To calculate the frequency of occurrence, only 238 stomachs were considered for *A. boyeri* and 355 for *A. presbyter*, the remained stomachs contained only prey completely digested, i.e., unidentified prey (table I).

Both species consumed a considerable variety of food items from different groups: worms (Annelida), molluscs (Bivalvia and Gastropoda), insects, crustaceans (Cumacea, Ostracoda, Copepoda, Isopoda, Amphipoda, Mysidacea and Decapoda), some fish, plants, detritus and eggs. The crustaceans were the dominant group, with a large variety of food items, representing almost 55% and 47% of total ingested food for *A. boyeri* and *A. presbyter*, respectively.

The small benthonic crustaceans, which comprise Amphipoda and Isopoda, were the most representative group in both species (fig. 3A and 3B), especially due to the presence of *Sphaeroma sp.* (Isopoda) (32 % of frequency of occurrence for *A. boyeri* and 22 % of frequency of occurrence for *A. presbyter*) (table I).

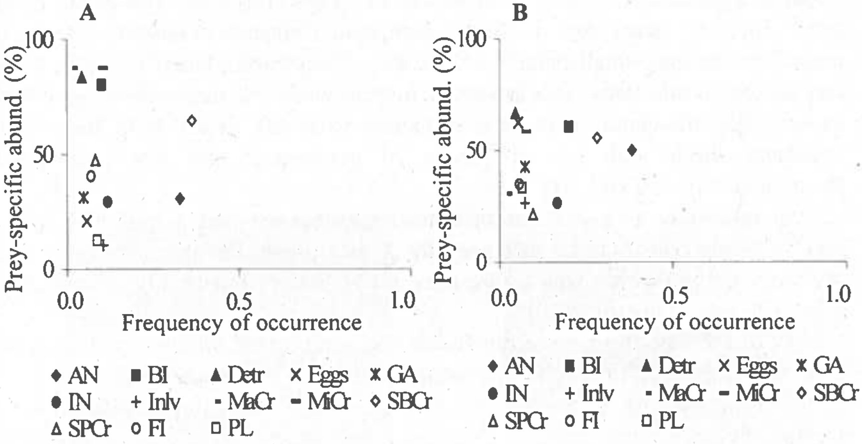

Figure 3. Prey-specific abundance per frequency of ocurrence (modified Costello method) (AMUNDSEN *et al.*, 1996) of **A** - *A. boyeri* and **B** - *A. presbyter*. AN - ANNELIDA; BI - BIVALVIA; Detr - Detritus; Eggs - Eggs; GA - GASTROPODA; IN - INSECTA; INlv - INSECTA larvae; MaCr - Macrocrustacea; MiCr - Microcrustacea; SBCr - Small benthonic crustacea; SPCr - Small pelagic crustacea; FI - Fishes; PL - Plants.

Annelids showed a different representativeness for the studied species. To *A. boyeri*, this prey composes a generalised item (fig. 3A), i.e. the prey was occasionally consumed by a large number of individuals, leading to a low prey-specific abundance, while to *A. presbyter* it constitutes a specialised prey (fig. 3B), i.e. it was characteristic of the diet of this species, leading to a high frequency of occurrence. Some specimens of *A. boyeri* tended to specialize the food strategy in the small benthonic crustaceans, with relevance to the Amphipoda (56 % of prey-specific abundance) (table I).

For *A. boyeri*, the items, which highly contributed to the width of their ecological niches (High BPC (between-phenotype component to niche width), were bivalves, microcrustaceans, macrocrustaceans and detritus (fig. 3A). On the other hand, for *A. presbyter*, the food items that contributed for a high BPC were microcrustaceans, detritus and eggs (fig. 3B). The item that highly contributed for the high BPC, for *A. boyeri* was the macrocrustacean group (fig. 3A) (with relevance to *Crangon crangon*, with 78 % of prey-specific abundance) (table I). For *A. presbyter* the item with more contribution was detritus (fig. 3B) with 77 % of prey-specific abundance) (tab. I).

The great variety of food consumed by these two species is largely due to the presence of many rare prey, which can indicate that, in the absence of their preferential items, these species feed on every type of available resources. The group of rare prey for *A. boyeri* include gastropods, insects, larvae of insects, small pelagic crustaceans, fishes, plants and eggs (fig. 3A). The group of rare prey for *A. presbyter* include gastropods, insects, larvae of insects, macrocrustaceans, small pelagic crustaceans, fishes and plants (fig. 3B). Within the group of rare prey, the larvae of insects were the rarest food item for *A. boyeri*, and the small pelagic crustaceans were the rarest food item for *A. presbyter*, both with low frequency of occurrence and low prey-specific abundance (fig. 3A and 3B).

It is important to point out that macrocrustaceans had a high BPC for *A. boyeri*, while constituted a rare prey for *A. presbyter*. The opposite occurs with the eggs group, which was a rare prey for *A. boyeri* but highly contributed to high BPC for *A. presbyter*.

In both species, there were not found any food items, which could contribute to a high WPC (within-phenotype component) or any dominant prey.

A comparison of niche breadth was achieved, the values obtained in the Levins measure were wide, (7.24, for *A. boyeri* and 8.99, for *A. presbyter*), reflecting the generalised feeding strategy in both species. The standardized Levins measure was relatively low, indicating reliance on a limited group of prey species, being 0.284 and 0.242 for *A. boyeri* and for *A. presbyter*, respectively (table II).

Table II. Levins' Measure (B) and Levins' Standardized niche breadth (B_A) for *A. boyeri* and *A. presbyter*.

Species	*A. boyeri*	*A. presbyter*
Levins' Measure (B)	7.24	8.99
Levins' standardized niche breadth (B)	0.284	0.242

Variation of food preference along the age

From the total 346 analysed stomachs of *A. boyeri*, 46 were from the 0^+-age group, 55 from the 1^+-group, 94 from the 2^+-group, and 43 from the 3^+-group From the total 493 analysed stomachs of *A. presbyter*, 43 were from the 0^+-age group, 105 from the 1^+-group, 180 from the 2^+-group, and 27 from the 3^+-group (table III). The total 3 empty stomachs (1 for *A. boyeri* and 2 for *A. presbyter*) belonged to the 0^+-age group.

In general way, the majority of food items were represented in all the age groups, in both species, in spite of their abundance being higher or lower in the different stages of age (fig 4 A and B). However, there are some food items, which were only consumed by a certain age group. Within microcrustaceans, Cumacea was only evident in the 0^+-group (table III), so it consists in a very small prey that only satisfies small predators. The opposite occurs with macrocrustaceans, such as *Crangon crangon* and other decapods that were mostly consumed by older individuals (mainly 2^+ and 3^+) (table III). *Praunus flexuosus* (Mysidacea) was only consumed by the 2^+-age group, in both studied species (table III). It is important to point out that the 2^+-age group, in both species, showed the largest variety of food items, comprising all the considered food groups (fig 4 A and B) and in the majority of the crustaceans subgroups (table III).

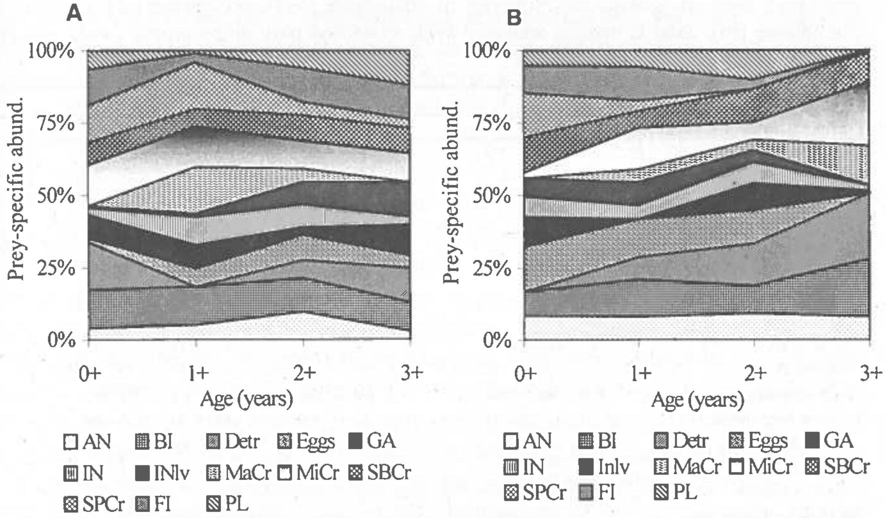

Figure 4. Prey-specific abundance of **A** - *A. boyeri* and **B** - *A. presbyter*, along the age (0^+, 1^+, 2^+, 3^+). AN - ANNELIDA; BI - BIVALVIA; Detr - Detritus; Eggs - Eggs; GA - GASTROPODA; IN - INSECTA; INlv - INSECTA larvae; MaCr - Macrocrustacea; MiCr - Microcrustacea; SBCr - Small benthonic crustacea; SPCr - Small pelagic crustacea; FI - Fishes; PL - Plants.

Comparing the diet of the two species, some food items were differently consumed along the age. Larvae of insects, fishes and eggs evidenced an opposite preference in each species. *A. boyeri* presented a high prey-specific abundance in the older stages of age (2^+ and 3^+-age group), while *A. presbyter* presented a high prey-specific abundance in the first years of age (0^+ and 1^+-age group) (fig. 4 A and B). Macrocrustaceans and detritus exhibited the opposite of the latter groups, the prey-specific abundance was higher in the first stages of *A. boyeri* and in the older stages of *A. presbyter*. Gastropods evidenced a large prey-specific abundance in different groups of age in each species; *A. boyeri* mostly consumed this item in the 1^+ and 3^+-groups, while *A. presbyter* mostly consumed this item in 0^+ and 2^+-groups.

Small pelagic crustaceans represented the highest prey-specific abundance (100 %) in the 1^+-group for *A. boyeri*, persisting along all the age groups. This percentage shows that this prey occurred in all the analysed stomachs for this group of age. For *A. presbyter* this food item showed the highest prey-specific abundance (100 %) in 0^+-group, reducing along the age, not occurring in the 2+-age group.

Table III. List of subgroups of identified crustaceans per age group for *A. boyeri* and *A. presbyter*, with percentage of frequency of occurrence (FO), percentage of prey-specific abundance (Pi), total stomachs analysed with identified prey and stomachs analysed per age.

Prey	0^+				1^+				2^+				3^+			
Phyllum Crustacea	Aboy		Apre		Aboy		Apre		Aboy		Apre		Aboy		Apre	
	FO	Pi	FO	Pi	FO	Pi	FO	Pi	FO	Pi	FO	Pi	FO	Pi	FO	Pi
Microcrustacea																
Order Cumacea	2.2	31.6	4.7	7.5												
Class Ostracoda	2.2	10.5			1.8	16.0	1.9	33.3	2.1	33.3	2.2	35.6				
Class Copepoda	8.7	90.9	2.3	20.0	9.1	80.9	3.8	89.2	1.1	81.8	3.3	24.1	2.3	76.9	3.7	98.0
Crustacea larvae											0.6	33.3				
Small benthonic crustacea																
Order Isopoda			2.3	25.0					1.1	11.1	1.1	33.3				
Gnatia sp.			2.3	100.0							0.6	4.8				
Idotea sp.					1.8	50.0	1.9	2.5	2.1	46.2	1.1	42.9				
Sphaeroma sp.	32.6	44.4	25.6	90.3	25.5	31.9	15.2	38.6	25.5	56.2	24.4	52.6	37.2	64.3	18.5	60.8
Order Amphipoda	2.2	11.1			5.5	66.7	1.9	3.1	1.1	33.3	6.7	47.3	2.3	70.0	7.4	12.9
Ampithoe sp.											0.6	20.0				
Corophium sp.											0.6	3.8				
Gammarus sp.											1.1	35.3				
Small pelagic crustacea																
Order Mysidacea	10.9	36.8	2.3	25.0	1.8	33.3	9.5	18.5	5.3	18.5	10.0	18.1	11.6	23.9	7.4	3.4
Gastrosaccus sp.											0.6	100.0				
Neomysis sp.											0.6	20.0				
Praunus flexuosus									1.1	33.3	0.6	50.0				
Macrocrustacea																
Order Decapoda											0.6	7.1			3.7	55.6
Crangon crangon					1.8	100.0	1.0	28.6	1.1	33.3	1.1	37.5			3.7	100.0
Total stomachs with identified prey	238		355													
0^+-age group	46		43													
1^+-age group	55		105													
2^+-age group	94		180													
3^+-age group	43		27													

DISCUSSION

Most of dietary components of *A. boyeri* and *A. presbyter* are pelagic but also benthonic organisms, as annelids, small benthonic crustaceans (amphipods and isopods), and molluscs (gastropods and bivalves). The apparent dominance of crustaceans in stomachs of fish reflects the preponderance of this taxon in the plankton, as TURNPENNY *et al.* (1981) verified for *A. presbyter*. For *A. boyeri*, crustaceans represented 55% of its food composition, while for *A. presbyter* represented 47%. Within the benthonic species, it is important to point out that annelids occupied the second place in both atherinids diet (32% for *A. boyeri* and 38% for *A. presbyter*). This group exhibited a distinct representativeness for each studied species, leading to a generalised food item for *A. boyeri* and showing a tendency for specialization for *A. presbyter*.

According to MARSHALL (1971), the atherinids show adaptations for feeding on the surface film. This behaviour was verified by GARCIA (1994) and also in the present study, where insects constitute the third place in the diet of both species (21 % for *A. boyeri* and 25 % for *A. presbyter*).

Previously, on account of the anatomy of fish, atherinids have generally been considered as planktivorous. In contrast, ROSECCHI & CRIVELLI (1992), for *A. boyeri*, and TURNPENNY *et al.* (1981), for *A. presbyter*, reported a predominantly zooplanktivorous diet. In this study, *A. boyeri* and *A. presbyter* were found microcarnivorous species (GARCIA, 1994; AZEVEDO, 1995), whose diet was based on small invertebrates. In terms of feeding habits, they are euryphagic (NIKOLSKY, 1963), as they feed on a large variety of prey, which leads to a generalist food strategy. Therefore, the degree of fullness was very high and the number of empty stomachs was very low, probably because the studied species, although primarily microcarnivorous, include in their diet various species of annelids, molluscs, macrocrustaceans, plants, fishes, eggs and detritus, which indicates a trophic plasticity, adapting it to use available food.

A comparison of niche breadth reflects the generalised feeding strategy in both species. This food strategy was clearly distinguished in both species by the tendency to a specialization for small benthonic crustaceans, for *A. boyeri*, which do not happen for *A. presbyter*. For each studied species, the food item that highly contributed to the width of their ecological niches (high BPC) was macrocrustaceans for *A. boyeri* and detritus for *A. presbyter*. This fact is according to FERNANDEZ-DELGADO *et al.*, 1988; CREECH, 1992 and LEONARDOS & SINIS, 2000, who evidenced that the length composition was markedly different in both species: *A. boyeri* was smaller but heavier than *A. presbyter*, which can probably be related to its food preference for big prey. Some other differences can distinguish this two species, like the fact of eggs highly contributed for the width of ecological niches for *A. presbyter*, while for *A. boyeri* this item showed a small contribution as it consisted in a rare prey.

Opposing the data of TURNPENNY *et al.* (1981), who proposed the absence of phytoplankton in *A. presbyter*, it is observed in this study, in both species, the presence of plants, although appearing with a low percentage of frequency of occurrence (8 and 7 % in the analysed stomachs for *A. boyeri* and *A. presbyter*, respectively).

In accordance with TURNPENNY *et al.* (1981), from the species found in stomachs and the minimum occurrence of phytoplankton, it is apparent that the two species does not graze the water body passively, but actively predates animals moving in the water, enabling it to select zooplankters from the general plankton, and with increasing age, to supplement its diet with larger swimming animals, such as amphipods and decapods. Cumacea, a small crustacean, was only consumed in the first year of life. Macrocrustaceans (decapods) with considerable dimensions were not consumed in the first year of life, but it was mostly evident in the older individuals. The fact of consumption of bigger prey as long as fish is growing is related with the size of fish mouth (BARBANÉ, 1980). ROSECCHI & CRIVELLI (1992) recorded a progressive decrease in the importance of plankton in the diet with increasing age of fish, which was not evident in this study.

It can be concluded that, in spite of both Atherinids species consumed the same food items, those items do not represent the same importance for both species, which is consistent with the existence of two species and not a synonymy. This fact improves the idea that they do not compete for the existing food, which certainly is largely available in the environment. Thus, it is possible to induce that the non-competition between these two sympatric species leads to a possible co-habitation in the same ecosystem, at the same time and space, as it occurs in the studied estuarine coastal lagoon, Ria de Aveiro.

ACKNOWLEDGEMENTS

We would like to thank the *University of Aveiro* and the *Foundation for Science and Technology* for their financial support of this research. The authors are also very grateful to Karine Briand and Alfredo Rocha for their field and lab collaboration.

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SOIL SPATIAL HETEROGENEITY IN MEDITERRANEAN ECOSYSTEMS

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Cruz, C., Muñoz, C., Botella, M. A. & Martins-Loução, M. A. (2002). Soil spatial heterogeneity in Mediterranean ecosystems. *Revista Biol. (Lisboa)* **20**: 73-80.

In the present study mineral nitrogen availability (nitrate and ammonium) and organic matter concentrations were determined for open space soils or for soils under the influence of several Mediterranean plant species (sclerophylous and summer semi-deciduous species).
The results obtained so far clearly show that: nitrate, ammonium and organic matter concentrations were quite variable within the research plot (macrogrid) even in the close proximity of individual plants. Variability in the sample values was great enough that some of the highest and lowest measured nutrient concentrations occurred within the extent of individual plant root systems. The spatial extent of variability was comparable to that found in similar ecosystems. According to the results obtained, it appears that soil landscape might be considered a mosaic of profiles reflecting the occurrence of chemical characteristics of the ground cover vegetation.

Key words: Spatial heterogeneity, soil nitrogen availability.

Cruz, C., Muñoz, C., Botella, M. A. & Martins-Loução, M. A. (2002). Heterogeneidade espacial do solo em ecossistemas Mediterrâneos. *Revista Biol. (Lisboa)* **20**: 73-80.

No presente trabalho estudou-se a heterogeneidade espacial de algumas características do solo: concentração de nitrato, amónio e matéria orgânica em solos sobre a influência das raízes e copas de plantas Mediterrâneas (espécies esclerófilas e semi-deciduas de Verão).
Os resultados obtidos mostram que a disponibilidade de azoto

mineral (nitrato e amónio) e a concentração de matéria orgânica variaram consideravelmente ao longo da área de amostragem, mesmo na proximidade dos indivíduos. A variabilidade dos parâmetros amostrados foi elevada permitindo que os valores máximos e mínimos registados se encontrassem dentro da área de um mesmo sistema radicular. A extensão espacial da variabilidade foi comparável à encontrada em estudos feitos para ecossistemas semelhantes. A integração dos resultados obtidos permite sugerir que o solo pode ser considerado como um mosaico que reflecte as características químicas do coberto vegetal.

Palavras chave: heterogeneidade espacial, disponibilidade do azoto no solo.

INTRODUCTION

Spatial soil heterogeneity is an inherent feature of virtually all plant communities, it has been suggested as a contributor to coexistence of plant species (MORRIS, 1999) and is also thought to affect the composition of plant communities (HUSTON & DEANGELES, 1994). Differences in the spatial and temporal distribution of nutrient transformation and consumption processes result in soil patches of differing concentrations, and the resistance to diffusion and mass flow between regions of high and low concentrations contributes to maintenance of patches (JACKSON & CALDWELL, 1996). The nutrient status of soil patches is affected by interactions of a variety of factors including plant cover (MORRIS, 1999), plant species composition (JACKSON & CALDWELL, 1993), microbial activity (SMITH *et al.*, 1994), soil temperature and moisture content. Since factors with the greatest influence on soil nutrients may vary over the growing season, the nutrient pool size of soil patches may change depending on which factors are most significant at a particular time (JACKSON & CALDWELL, 1996).

In most-temperate region ecosystems nitrogen is the nutrient that limits plant growth, and thus is the nutrient most likely to affect community structure if its availability is patterned (MORRIS, 1999).

If plants are important in regulating nutrient availability or distribution in soils, it would be expected that plants belonging to different functional types would contribute differently to the dynamics of nutrient availability in the soil. Most of the shrubs and trees found in Mediterranean ecosystems belong to one of two functional types: summer semi-deciduous and sclerophylous species.

Semideciduous species exhibit leaf dimorphism and have several survival mechanisms: shedding an important fraction of leaves and twigs in the summer - a stress avoidance strategy; production of thicker leaves better able to survive summer drought, with a more efficient stomatal control - stress tolerant strategies (CORREIA, 1988); production of short-lived leaves. Their development is very dependent on water availability in the upper soil. They are common in open and disturbed stands, but they are progressively eliminated under canopies of evergreen sclerophyllous species of the late successional stages (CORREIA & CATARINO, 1994).

Sclerophyll leaves are long-lived, consistent, hard and coriaceous, and break when folded. Plants pay a high construction cost for leaf protective structures and characteristics such as: thick cuticles; sclerits in the mesophyll; grouping of fibres into hypodermal layers; lignified epidermal cells; increased parenchymatous cell wall thickness; or higher cellulose content of parenchymatous cells (CORREIA & CATARINO, 1994). Furthermore, sclerophyllous plants are able to develop deeper root systems and to maintain a positive carbon balance throughout the year, though with reduced photosynthetic rates during summer (WERNER *et al.*, 2000).

In this study we assess the spatial variability of three soil resources essential for plant growth: ammonium, nitrate and organic matter and we super-impose the distribution of their spatial heterogeneity with the spatial distribution of plants according to their functional group.

MATERIALS AND METHODS

Studies were carried out in a mixed sclerophyll scrub. The revegetation was spontaneous and started after fire, 16 years ago. The site is situated in Serra da Arrábida, 50 km south of Lisbon (38°27'34''N, 9°0'20''W), on a south facing slope of Jaspe Peak, a calcareous elevation with 270 m. The climate is classified as sub-humid, warm variant according to Emberger's pluviometric coefficient, with an average annual precipitation of 650 mm, and 16°C mean annual temperature (CORREIA & CATARINO, 1994). The soil is very thin (maximum of 20 cm depth) and has a calcareous origin. After destruction of organic matter with hydrogen peroxide, soil texture was analysed using a nest of sieves with 212 and 63 µm mesh. Results showed that soil was formed by 15.1% sand, 56.7% lime and 28.2% clay. Values of air temperature (14°C), air relative humidity (78%), light intensity (600 μmol m^{-2} s^{-1}), soil temperature (13.2°C) and soil moisture content (0.121 m^3 m^{-3} soil) were determined *in situ* using a TDR probe (Delta Instruments) at the time the samples were taken. To study soil spatial variability at a medium scale the studying area, 2500 m^2, was divided in 25 squares of 100 m^2 each, each square was divided again in 25 squares of 4 m^2 each, giving a total of 625 sampling points. Samples were taken from the square centres and consisted of three cores of 2 cm in diameter and 15 cm length taken within 1 cm of each other and at 20 cm from the main stem of the nearest plant. Samples were stored at 4°C for analysis. Sampling took place in November 2000, before the heavy rain that occurred later in the month.

Sieved (2 mm) soil samples were analysed for: organic matter, by loss of ignition. Nitrate and ammonium were extracted from the soil using 2M KCl in a proportion of 1 g soil dry weight to 10 ml of extractant. Nitrate concentration was determined by electrophilic substitution of salycilate acid; and ammonium concentration using the Berthelot reaction (CRUZ & MARTINS-LOUÇÃO, 2000). The values obtained for organic matter were expressed as percentage of dry (80°C) soil and were grouped in 5 classes: 0-5, 5-10, 10-15, 15-20 and >20%. Nitrate and ammonium concentrations were expressed in μgg^{-1} and were grouped in 4 classes: 0-5, 5-10, 10-15 and 15-20 gg^{-1}.

Simultaneously a detailed map of the vegetation was done, using the same sampling net and considering the dominant species in each 4 m^2 cell. The species found were classified as summer-semideciduous or as sclerophyllous. The summer semideciduous species found were: *Cistus albidus, Cistus salvifolius, Cistus ladanifer, Cistus mospeliensis, Lavandula* sp., and *Rosmarinus officinalis*. The sclerophylous species were: *Ceratonia siliqua, Erica* sp., *Juniperus phoenicea, Myrtus communis, Olea europaea, Phillyrea angustipholia, Pistacea lentiscus, Rhamuns alaternus, Quercus coccifera*, and *Arbutus unedo*.

RESULTS AND DISCUSSION

The work here presented is part of a more ambitious project intended to describe nitrogen availability to plants over space and time. This information combined with the phenological characteristics of the dominant species (OLIVEIRA, 1995; WERNER, 2000) will provide information on nitrogen availability on Mediterranean ecosystems.

The main plant species found in the sampling area were grouped in summer semi-deciduous and sclerophylous species. The distribution of these two functional groups is represented in Fig.1. It was obvious that the vegetation was dominated by the sclerophylous species. The summer semi-deciduous species formed patches and *Cistus albidus* was the most abundant species. The other studied parameters (organic matter, nitrate and ammonium concentrations) were analysed and compared with distribution of the plant functional groups.

Generally, nitrate (Fig. 2), ammonium (Fig. 3) and organic matter (Fig. 4) concentrations were quite variable within the research plot (macrogrid) even in close proximity to individual plants. Variability in the sample values was great enough that some of the highest and lowest measured nutrient concentrations occurred within the extent of individual plant root systems. The spatial extent of variability was comparable to that found by JACKSON & CALDWELL (1993) and CAIN *et al.* (1999) in similar ecosystems. No significant correlation was found between nitrate and ammonium ($r^2 = 0.1687$), or inorganic nitrogen and organic matter ($r^2 = 0.0937$) concentration in the soil.

Total nitrogen concentration in Mediterranean soils is, generally, very low (CALDWELL 1994, CAIN *et al.*, 1999) and highly heterogeneous (ROBERTSON *et al.*, 1997), and data on nitrogen availability are difficult to obtain. Most of the 625 samples analysed had nitrate concentrations between 0 and 10 μgg^{-1} (Fig. 2). The most impressive result of the comparison between plant distribution and nitrate concentration patterns (Fig. 1 and 2) is that high nitrate concentrations were always associated with sclerophyllous vegetation. In most of the samples

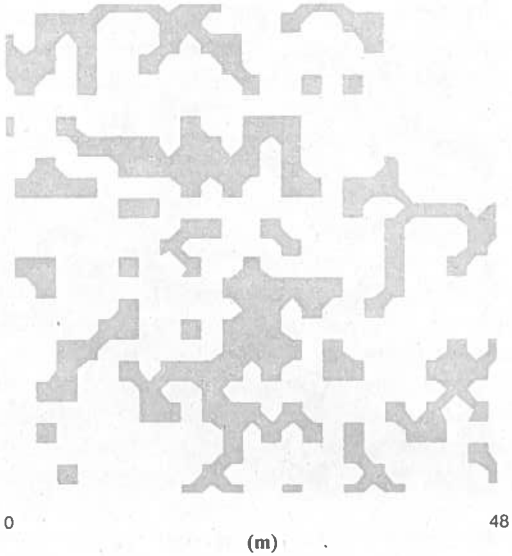

Figure 1 - Distribution of the plant species according to the respective plant functional groups: □ summer semi-deciduous species, and ▨ sclerophyllous species.

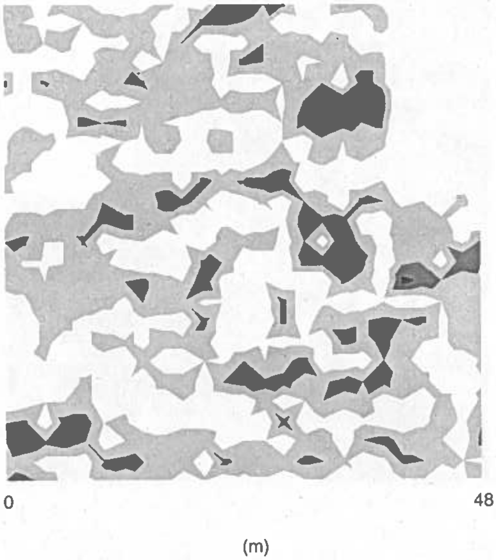

Figure 2 - Representation of the spatial heterogeneity of soil nitrate concentration: □ 0-5, □ 5-10, ▨ 10-15, and ■ 15-20 μgg^{-1}.

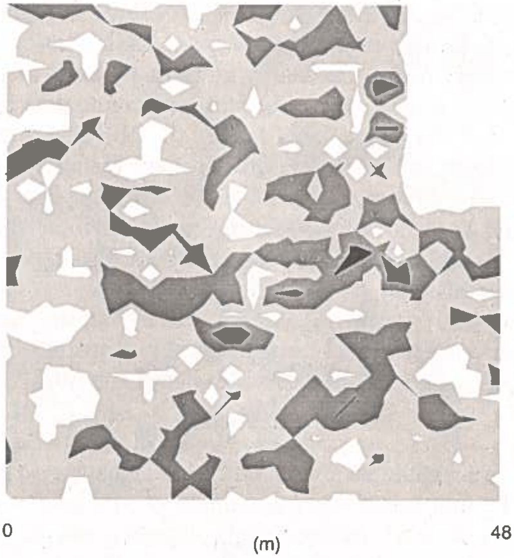

Figure 3 - Representation of the spatial heterogeneity of soil ammonium concentration:

□ 0-5, ░ 5-10, ▓ 10-15, and ■ 15-20 μgg^{-1}.

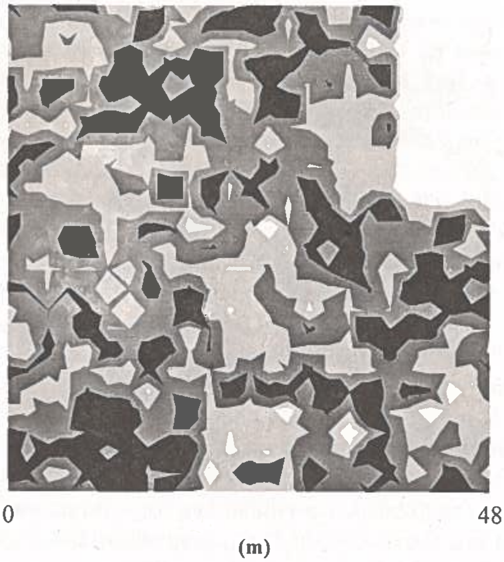

Figure 4 - Representation of the spatial heterogeneity of soil organic matter:

□ 0-5, ░ 5-10, ▓ 10-15, and ■ 15-20%.

ammonium concentration (Fig. 3) was higher than that of nitrate (Fig. 2) and no evident relation was observed between the pattern of ammonium distribution in the soil and that of the plant vegetation distribution. This might be related with other variables, such as soil temperature and water in the soil, that are very strictly correlated with the microbiological activity in the soil (ROBERTSON *et al.*, 1979).

Organic matter concentration in the soil was very heterogeneous (Fig. 4). High values for organic matter content were always associated with sclerophyllous plants (Fig. 1), mainly those of larger stem diameters (results not shown). This result may at least in part, be explained by the structure of the plants. Most of the sclerophyllous species here present develop relatively big canopies that at some points touch the soil and form a micro-environment more protected from wind, with higher water availability and less changeable soil temperatures allowing the decomposition of the shedded leaves and an higher turnover of the fine roots (GROSS *et al.*, 1995). The dominant summer semideciduous species (*Cistus albidus*) is a small shrub with straight stems developing an incipient canopy that does not aggregate the shedded biomass.

ROBERTSON *et al.* (1997) showed that soil pH, organic matter content, and assorted mineral element concentrations may vary in some communities by an order of magnitude at spatial scales of 5 m or less, and be associated with plant species distributions. Such results suggest that the nutrient cycling properties in natural and recently disturbed systems are spatially complex, and moreover that this complexity may significantly affect plant community structure (RYEL *et al.*, 1996). The co-existence of plant species at the study site highlights the possibility that variations in soil characteristics leading to variations in competitive abilities may be one of the mechanisms responsible for the dynamic of specific populations and processes occurring at the ecosystem scale.

CONCLUSIONS

Although preliminary, the results clear show a relation between the amount of nitrate and organic matter in the soil and the plant functional group to which the nearest plant belongs. Nevertheless no correlation was found between nitrate and ammonium or inorganic nitrogen and organic matter concentrations in the soil.

ACKNOWLEDGEMENTS

Due to the complexity of the sampling procedure and to the high number of analyses involved in this work the authors would like to thank the collaboration of the students from FCUL: João Barroca, Nuno Guerra, Pedro Lima, Ana Clara Grosso, Margarida Matos, Hugo Evangelista, Helena Soares, Joana Silva, Marta Mesquita, Sandra Batalha, and to the technician Herculana Costa. The authors thank Dr. Stephen Houghton for manuscript revision and fieldwork.

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OS SIG NA ANÁLISE DA BIODIVERSIDADE EM AGROECOSSISTEMAS: APLICAÇÃO NA REGIÃO ENVOLVENTE DE ÉVORA

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Batista, M. T. F. (2002). Os SIG na Análise da Biodiversidade em Agroecossistemas: Aplicação na Região Envolvente de Évora. *Revista Biol. (Lisboa)* **20**: 81-95.

Apresentam-se os resultados da análise da biodiversidade em agroecossistemas utilizando um Sistema de Informação Geográfica (SIG). A biodiversidade, caracterizada através do índice de diversidade de Shannon (H'), é relacionada numa perspectiva integrada, com a disponibilidade de recursos (alimento, abrigo e água) e factores de perturbação. Averigua-se em treze áreas de pormenor, a relação entre a diversidade faunística e a diversidade de *habitats*, a disponibilidade de água, a presença humana, a rede viária, o tipo de regime cinegético e o tipo de agroecossistema. Como resultado identificam-se algumas tendências espaciais, nomeadamente as áreas fragilizadas em termos de biodiversidade, aquelas onde existe maior quantidade de recursos e de perturbações observando-se as relações entre elas. Este estudo preliminar constitui a base para a monitorização da biodiversidade na região de Évora.

Palavras chave: Ecologia da Paisagem, agricultura, biodiversidade, SIG.

Batista, M. T. F. (2002). GIS Applied to Biodivertity Assessment on Agro Ecosystems: The Évora Case Study. *Revista Biol. (Lisboa)* **20**: 81-95.

This paper reports the results assessing biodiversity on agro ecosystems using GIS. Biodiversity is determined using Shannon diversity index (H') and related with resources availability (food, water and shelter) and disturbance factor of tranquility and responsible for its diminution. We analyzed the relations between fauna diversity, habitat diversity, water availability, human settlements, roads, hunting regime and agro ecosystems typology in thirteen detail areas. As results we identify some spatial tendencies, such as stressed areas concerning diversity, the geographic distribution of resources and disturbance factors and the relation between them. This preliminary study is the base for monitoring biodiversity in Évora region.

Keywords: Landscape Ecology, agriculture, biodiversity, GIS.

INTRODUÇÃO

A conservação da biodiversidade em agroecossistemas tem vindo a ser largamente discutida a nível da União Europeia, tendo sido incorporadas na Reforma da Política Agrícola Comum em 1992, e mais tarde, na Agenda 21, medidas agro-ambientais com o objectivo de limitar o impacte da actividade agro-pecuária na biodiversidade. Contudo a relação causa-efeito agroecossistema / biodiversidade é ainda insuficientemente conhecida, especialmente quando analisada numa perspectiva integrada: *biodiversidade / recursos / factores de perturbação*.

A paisagem da região envolvente de Évora, tem sofrido ao longo dos tempos sucessivas alterações resultantes quer da intensificação da agricultura tradicional extensiva de sequeiro, quer através da expansão do núcleo urbano de Évora e áreas periurbanas adjacentes. Estes e outros factores tais como a densidade da rede viária e o tipo de gestão cinegética afectam a biodiversidade local. O presente estudo, inserido na sequência de estudos já efectuados para a área envolvente de Évora[1], pretende avaliar a diversidade faunística e a sua relação com a distribuição geográfica dos recursos e factores de perturbação, utilizando os Sistemas de Informação Geográfica para a compatibilização e análise da informação. A relação entre a diversidade faunística e a diversidade de habitats, assenta no pressuposto de que a diversidade específica de uma dada região é, em parte, função da diversidade de habitats (COOPERRIDER, 1995).

Aplicam-se índices de diversidade e de dominância à área de estudo, os quais tem provado ser de grande utilidade, quer na biologia (MAGURRAN, 1988) quer na ecologia da paisagem, permitindo a caracterização da estrutura da paisagem e da sua evolução ao longo do espaço e do tempo (DRAMSTAD *et al*, 1998). A sua aplicação, embora continue a provocar controvérsia entre a comunidade científica, é largamente apetecível entre os planeadores que necessitam de instrumentos para identificar e monitorizar os impactes causados pela actividade humana na paisagem e biodiversidade. Este tipo de índices

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[1] Projecto "Análise da Região Envolvente de Évora numa perspectiva da Ecologia da Paisagem" (Programa STRIDE), 1992-95, Centro de Estudos de Ecossistemas Mediterrânicos (C.E.E.M.), Universidade de Évora e projecto "Gestão da Informação na Planificação do Espaço Rural – Aplicação no Distrito de Évora" (Contrato 16/94 JNICT/DGOTDU), UNESUL- Associação Universidade Empresa do Sul de 1995-97).

permite, entre outras, a análise comparativa da diversidade de espécies e habitats presentes num determinado local e momento temporal (FORMAN & GODRON, 1986, FORMAN, 1995, McGARIGAL & MARKS, 1995).

METODOLOGIA

A área de estudo (AE) abrange 36100 hectares da área envolvente da cidade de Évora onde se encontram definidas treze áreas de pormenor no *core* das quais se efectuaram os levantamentos faunísticos (MASCARENHAS et al. 1995, RABAÇA et al. 1995, 1996, RAIMUNDO, 1995) (fig. 1).

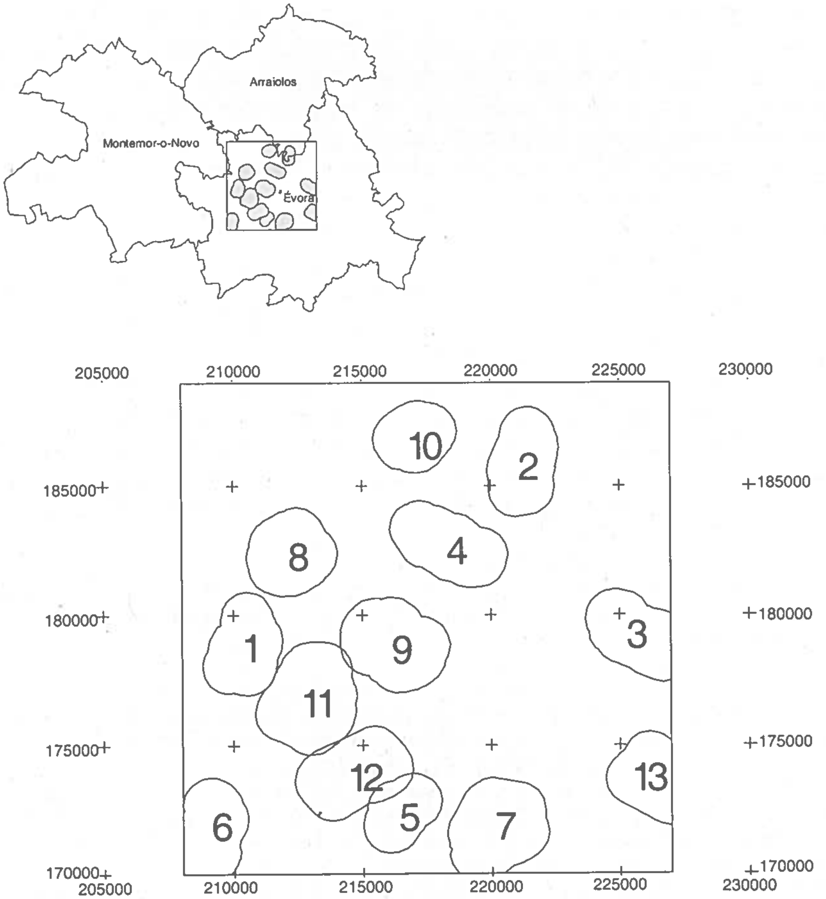

Figura 1 - Localização da área de estudo e das áreas de pormenor (Coordenadas Heyford-Gauss, Datum de Lisboa)

Os *habitats* dominantes são as culturas arvenses de sequeiro em terra campa (45%) e os montados em geral (35%: azinho 17%, sobro 7% e montado misto 11%). As Culturas arvenses de regadio, vinhas, pomares e policultura representam 12%, enquanto que o eucaliptal e o pinhal ocupam 2,5 %, sensivelmente o mesmo que o olival (2,2 %). A área urbana contínua e descontínua ocupa 2,8 % enquanto os matos e folhosas arbustivas têm pouca expressão (0,3%). A tabela 1 caracteriza as treze áreas de pormenor quanto à abundância relativa de *habitats*.

A análise efectuada baseia-se no modelo funcional do *habitat* para a fauna, em termos de recursos/necessidades: Alimento – Abrigo – Água (os 3 Ás). São analisados como recursos a estrutura da paisagem (constituindo *habitats* de alimentação e de abrigo) e a disponibilidade de água. Como perturbações analisam-se a existência de uma rede de vias (perturbação da tranquilidade e mortalidade por atropelamento), a presença humana em áreas de tecido urbano contínuo (TUC), tecido urbano descontínuo (TUD) e equipamentos (perturbação da tranquilidade e factor potenciador de espécies oportunistas), o tipo de agroecossistema (quanto ao grau de intensificação da agricultura definido pelo grau de utilização de maquinaria pesada, de fitofármacos e fertilizantes, o encabeçamento pecuário, etc.) e a sujeição a regime cinegético geral ou especial (caça controlada ou não). Os agroecossistemas da AE são tipificados de acordo com o tipo de exploração agrícola e orientação técnico-económica, tendo por base os inquéritos efectuado às explorações agrícolas entre 1993 e 1994 (MASCARENHAS, 1995). Identificaram-se dois Sistemas de Produção Agrícola Familiar (SPAF) e cinco Sistemas de Produção Agrícola Empresarial (SPAE), três de sequeiro e dois de regadio (MARQUES e FRAGOSO, 1998) (ver tabela 2).

A estrutura da paisagem caracteriza-se através do índice de diversidade de SHANNON (1962)[2], a heterogeneidade pelo índice de SIMPSON modificado proposto por ROMME & KNIGHT (1982)[3] e pelo índice do padrão de organização proposto por O'NEILL *et al.* (1988)[4]. Embora estes índices

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[2] O índice de Shannon (1962) (H') é um índice quantitativo que permite medir a diversidade de habitats, relacionando o número de tipos de habitats com a área por eles ocupada relativamente à totalidade da área. $H' = - \sum pi\ ln\ pi$, em que p*i* é a proporção da área total do habitat coberta pela categoria *i* de coberto vegetal.

[3] A complexidade e heterogeneidade da paisagem, isto é o padrão de distribuição dos habitats e o seu grau de fragmentação são dados pela semelhança da paisagem, que mede as proporções relativas de representação aérea entre tipos de vegetação numa paisagem, e é determinado usando o índice de Simpson modificado, proposto por Romme e Knight (1982) (E = H2 / ln T, onde H2 = -ln ∑ pi2, onde T – é o número dos diferentes tipos de comunidades presentes e pi – é a proporção da paisagem total coberta por uma comunidade). O valor deste índice aumenta quando a cobertura aérea relativa dos diferentes tipos de vegetação tem um padrão de distribuição homogéneo, e decresce com o aumento da dominância aérea de um tipo de vegetação.

[4] O índice do padrão de organização proposto por O'Neill *et al.* (1988) (D1), permite medir a extensão em que alguns tipos de uso do solo dominam a paisagem. $DI = ln\ n + \sum pi\ ln\ pi$, onde *pi* – é a proporção de uma unidade de uso i na área de estudo, *n* – número total de categorias de uso existentes na área de estudo, *ln n* – representa um máximo, com todos os tipos de uso presentes em proporções iguais.

Tabela 1 - Habitats das áreas de Pormenor (APs) em hectares.

HABITAT	AP1	AP2	AP3	AP4	AP5	AP6	AP7	AP8	AP9	AP10	AP11	AP12	AP13	AETotal
Superficie (ha)	906,47	937,13	849,56	1050,85	670,42	799,41	1166,29	910,23	1162,23	686,95	1278,49	1134,91	671,30	36100,46
Habitats (% da sup. da AP)														
Montado de Azinho	4,52	51,57	1,46	1,39	33,05	13,13	7,01	13,72	10,21	17,29	36,43	65,24	0,02	16,60
Montado de Sobro	19,47	10,46	44,22	0,89	-	4,13	-	13,50	18,70	-	9,22	-	-	7,30
Montado misto	64,87	-	-	0,41	-	18,89	0,06	56,05	23,00	-	36,70	-	-	11,15
Olival	4,18	0,29	0,43	3,19	3,60	0,39	1,66	0,00	2,31	-	4,49	9,85	-	2,21
Culturas arvenses de sequeiro em terra campa	5,28	37,60	29,30	10,84	53,74	21,95	88,03	14,16	7,05	77,32	9,68	11,33	99,50	45,05
Culturas arvenses de regadio, vinhas, pomares e policultura	1,05	-	16,17	83,00	-	9,74	1,17	1,16	29,85	4,73	2,52	0,37	-	11,98
Tecido urbano descontínuo + Áreas sociais	0,55	0,09	0,06	0,11	5,65	8,99	1,18	0,43	6,00	0,16	0,13	4,61	0,18	2,86
Pinhal e eucaplital	0,08	-	8,36	0,16	1,21	22,79	0,91	0,22	0,18	0,49	0,84	7,66	0,31	2,56
Matos e folhosas arbustivas	-	-	-	-	2,77	-	-	0,60	2,70	-	-	0,94	-	0,31
TOTAL	100	100	100	100	100	100	100	100	100	100	100	100	100	100

Tabela 2 - Caracterização dos Agroecossistemas da Região envolvente de Évora

				SAU	AF	O	B	CN	SEQ	REG	P	FTO		ADU		MAQ		INTS	AGRSS
TIPO DE AGROECOSSISTEMA	ORIENTAÇÃO TÉCNICO-ECONÓMICA	PRINCIPAIS TIPOS DE PRODUÇÃO	UTA	Área média por exploração(ha) S.A.U.	Superf. Forrageira (ha)	Ovinos	Bovinos	Encabeçamento (CN/ha)	cereais e oleaginosas de sequeiro (ha)	cereais e oleaginosas de regadio (ha)	a s s e i t d - e	Utilização de fitofármacos		Utilização de adubos		Utilização de maquinaria pesada		Grau de intensificação *	agressividade
												Aspectos técnicos	Valor (0-3)	Aspectos técnicos	Valor (0-3)	Aspectos técnicos	Valor (0-3)		
Agro 1	SPAF	culturas forrageiras e ovinos	1	10	10	50	0	0,75	0	0	0	não se monda	0	1 adubação	1	escarif.+ grad	1	0,75	1
Agro 2	SPAF	culturas forrageiras, ovinos e bovinos de carne	1	25	25	20	6	0,36	0	0	0	não se monda	0	1 adubação	1	escarif.+ grad	1	0,36	1
Agro 3	SPAE de sequeiro	culturas forrageiras e bovinos de carne	5	720	720	0	350	0,49	0	0	0	não se monda	0	1 adubação	1	escarif.+ grad	1	0,49	1
Agro 4	SPAE de sequeiro	culturas forrageiras, cereais e oleaginosas e bovinos de carne	4	590	418	0	280	0,67	150	0	22	1 monda	1	2 adubações	2	Charrua + gradagem	2	5,64	2
Agro 5	SPAE de sequeiro	culturas forrageiras, cereais e oleaginosas, ovinos e bovinos de carne	7	1170	550	300	280	0,59	450	0	68	1 monda	1	2 adubações	2	Charrua + gradagem	2	5,58	2
Agro 6	SPAE de regadio	culturas forrageiras, cereais e oleaginosas de regadio, bovinos de carne	3	500	375	0	200	0,53	0	110	17	1 a 2 mondas	2	3 adubações	3	Charrua + gradagem	2	7,47	3
Agro 7	SPAE de regadio	culturas forrageiras, cereais e oleaginosas de sequeiro e de regadio e bovinos de carne	7	850	520	0	270	0,52	100	200	30	1 a 2 mondas	2	3 adubações	3	Charrua + gradagem	2	7,50	3

* O grau de intensificação é calculado a partir da seguinte equação:

INTS=CN+((SEQ+REG)*FTO+(SEQ+REG)*ADU+(SEQ+REG)*MAQ)/((SEQ+REG)+1), tendo por base os dados dos inquéritos efectuados a 58 explorações agrícolas de grande dimensão da área de estudo, entre 1993-1994.

CN/ha= (0,15*ovinos+1*bovinos)/AF; UTA - Unidade trabalho ano; grad. - gradagem; escarif. - escarificação; S.A.U. - superfície agrícola útil

Tabela 3 - Caracterização das áreas de pormenor : diversidade faunística, recursos e factores de perturbação

	Áreas de Pormenor (APs)	AP1	AP2	AP3	AP4	AP5	AP6	AP7	AP8	AP9	AP10	AP11	AP12	AP13
Diversidade Faunística	Aves (H') (S=63)	3,01	3,01	3,16	2,98	3,19	3,50	1,95	2,79	2,83	2,57	3,04	2,89	2,10
	Répteis (Exp H') (S=11)	1,00	1,65	1,36	1,00	1,00	1,76	5,00	1,96	1,00	1,00	1,89	1,00	2,00
	Anfíbios Adultos (Exp. H') (S=10)	3,04	3,24	2,71	2,23	2,94	2,29	2,23	3,19	4,50	2,77	3,64	2,57	1,63
	Anfíbios Larvas (Exp. H')	1,48	2,52	2,00	1,89	1,38	1,00	2,00	1,96	3,41	2,24	3,36	1,21	1,00
Recursos	Diversidade de Habitats (Índice de Shannon H')	2,79	1,82	2,41	2,51	1,75	2,53	2,16	2,62	2,94	1,77	2,83	2,50	1,75
	Heterogeneidade (Índice Simpsom Modificado E)	0,84	0,77	0,75	0,73	0,66	0,83	0,82	0,75	0,77	0,77	0,84	0,73	0,95
	Padrão Organização (D1)	5,83	4,12	5,18	5,40	3,95	5,30	4,56	5,57	6,23	3,85	5,92	5,45	3,54
	Disponibilidade de Água	2	1	2	1	2	3	0	1	1	0	1	1	0
Factores de Perturbação	Densidade de Vias	2	0	0	3	0	1	1	1	3	2	1	1	1
	Presença Humana	0	0	1	3	0	0	0	0	2	0	1	0	0
	Tipo de Agroecossistema	2	2	0	0	2	3	2	1	1	2	2	2	2
	Regime Cinegético	3	3	2	0	3	3	3	3	0	3	3	3	2

apresentem elevada colinearidade considerou-se que a determinação única do índice de diversidade não elucidaria a distribuição geográfica dos diferentes *habitats*, o que nos é evidenciado pelo índice de SIMPSON e pelo padrão de distribuição das diferentes manchas de *habitats*.

O índice de SHANNON (H') é igualmente aplicado à caracterização da diversidade de aves (S=63), utilizando-se a Exponencial de H' no caso dos répteis e anfíbios devido à relativa pequena dimensão da amostra (S=10 e S=11 respectivamente) (ver tabela 3).

A disponibilidade de água e os factores de perturbação são valorizados numa escala de 0 a 3, de acordo com a sua abundância e padrão de distribuição, tal que:

Disponibilidade de água

Com base na distribuição da rede hidrográfica da Carta Militar de Portugal à escala 1:25000 do Instituto Geográfico do Exército (IGEOE) e charcos e pequenas barragens, valoriza-se de 0 a 3 a água disponível, onde 0 representa as zonas onde apenas existem linhas de água temporárias, 1 onde existem linhas de água permanente e alguns charcos de pequena dimensão, 2 onde abundam linhas de água permanente e charcos de pequena e média dimensão homogeneamente dispersos e finalmente, com valor 3, foi classificada a área de pormenor onde se encontra a Albufeira da Tourega a qual ocupa cerca de 10% da respectiva área de pormenor.

Densidade e distribuição da rede viária

A densidade de estradas e a sua distribuição na área de pormenor, classifica-se tal que 0 – inexistência de vias a atravessar a área; 1 – atravessado por uma via; 2 – atravessado por duas vias; 3 – atravessado por três ou mais vias.

Tipo de agroecossistema

Cruzando a Carta do Cadastro, onde se encontram identificadas as explorações agrícolas da AE, determina-se o tipo de agroecossistema dominante em cada área de pormenor. O grau de intensificação agrícola e pecuária define o seu grau de agressividade e tem por base as características dos sistemas de produção fornecidos nos ínquéritos aos agricultores efectuados em 1995-96 (ver tabela 2), onde: 0 – sistema de produção pouco agressivo; 1 – sistema de produção medianamente agressivo (agroecossistemas 1, 2 e 3); 2 – sistema de produção agressivo (agroecossistemas 4 e 5) e 3 – sistema de produção muito agressivo (agroecossistemas 6 e 7).

Caça controlada ou não

O cadastro cinegético permite determinar, para cada área de pormenor, a percentagem de área sob o Regime Cinegético Especial (RCE). Este factor valoriza a existência de caça controlada em oposição ao regime livre, uma vez que se considera a gestão de populações e controle da pressão cinegética elementos benéficos para a maioria das espécies, tal que: 0 – 0 a 10% da área sob regime de caça especial; 1 – 10 a 30% da área sob regime de caça especial; 2 – 30 a 50 % da área sob regime de caça especial; 3 – mais de 50% da área sob regime de caça especial.

Presença humana, densidade e padrão de distribuição

A presença de áreas urbanas contínuas (TUC), descontínuas (TUD) e de equipamentos influencia o tipo e número de espécies ocorrentes, quer pela perturbação causada pela movimentação, ruído e presença humana, quer pela fragmentação dos habitats, quer ainda pela influência que tem na dominância das espécies, favorecendo umas em detrimento de outras (generalistas em detrimento de especialistas). Este factor é valorizado de acordo com a percentagem da área de pormenor ocupada, tal que: 0 – 0 a 10%; 1 – 10 a 30%; 2 – 30 a 50 %; 3 – mais de 50%.

RESULTADOS E DISCUSSÃO

A tabela 3 apresenta os valores obtidos para cada área de pormenor relativamente aos factores analisados. As figuras 2 a 4 ilustram a distribuição dos factores analisados por área de pormenor, sendo a sua distribuição espacial representada nas figuras 5 e 6.

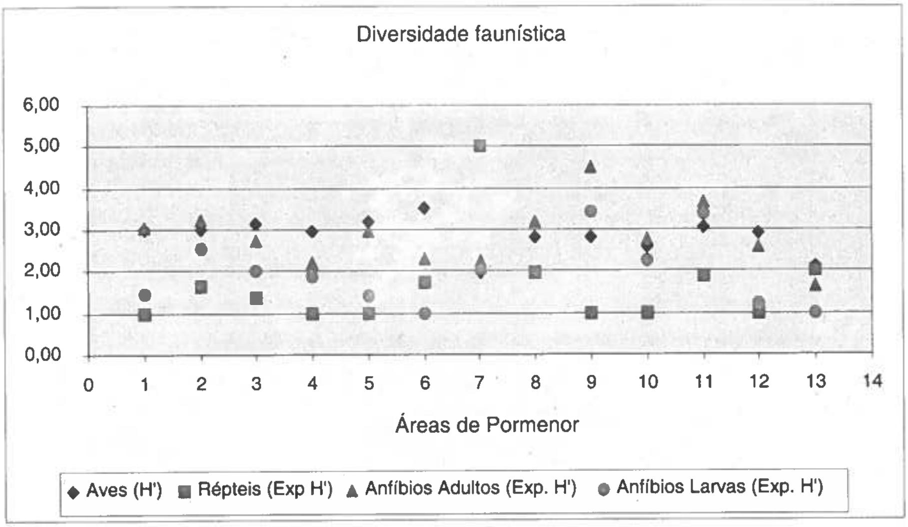

Figura 2 – Diversidade por grupo faunístico e área de pormenor

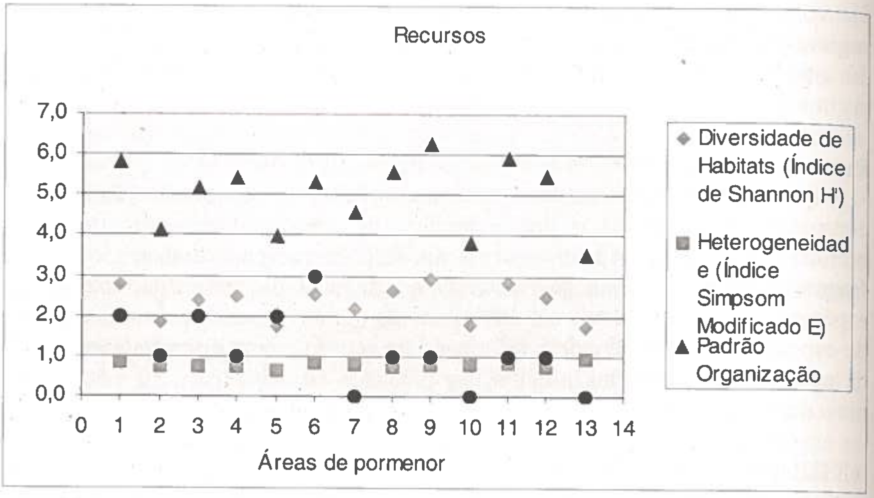

Figura 3 –Recursos por área de pormenor

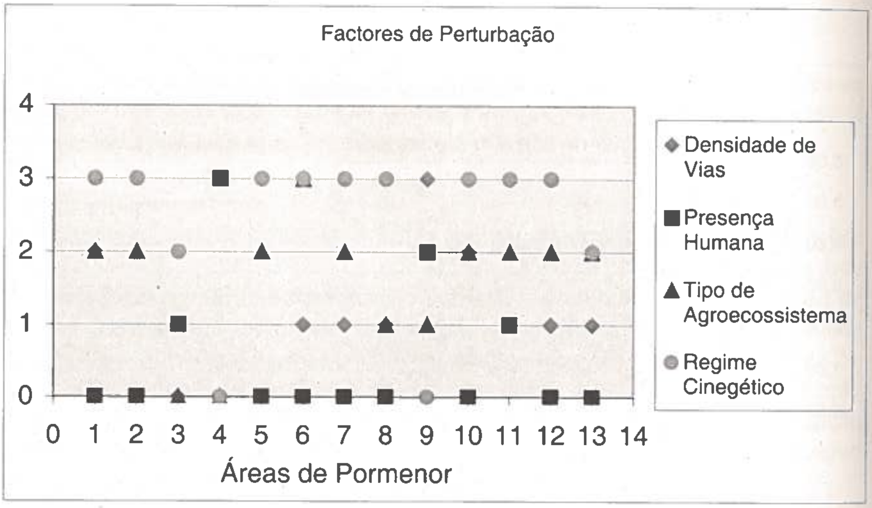

Figura 4 – Factores de perturbação por área de pormenor

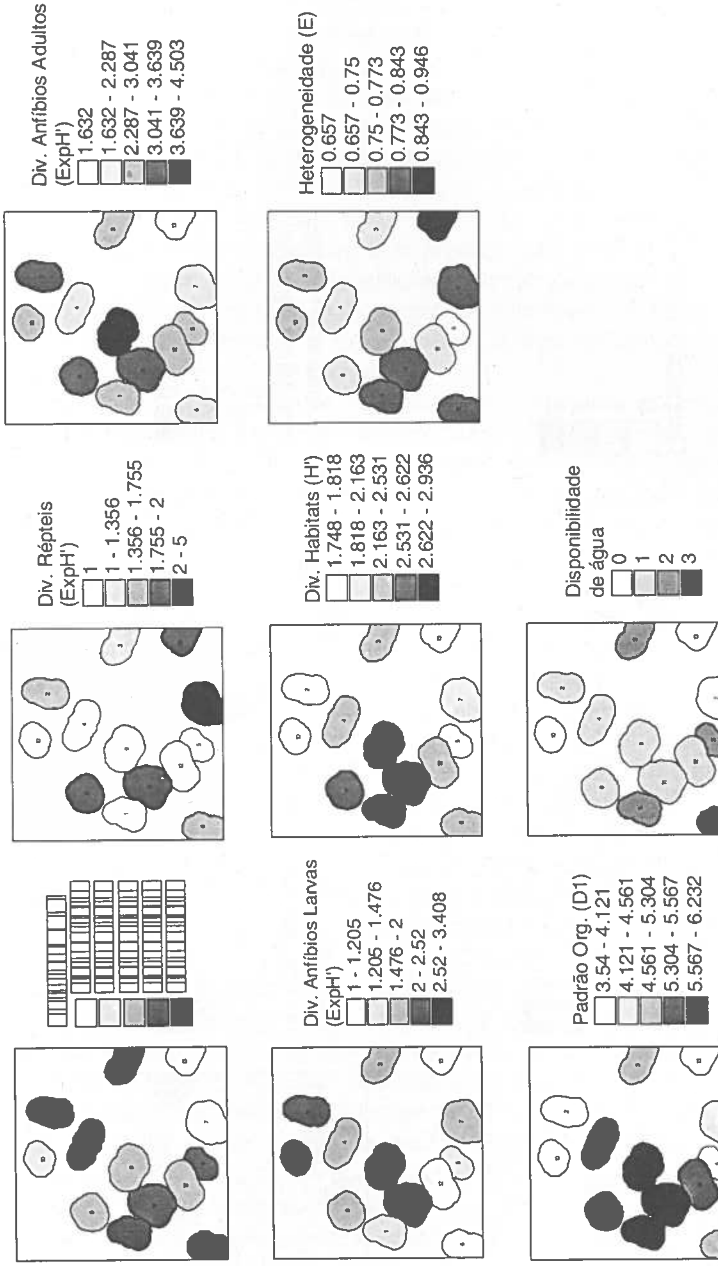

Figura 5 – Distribuição espacial da diversidade faunística e recursos

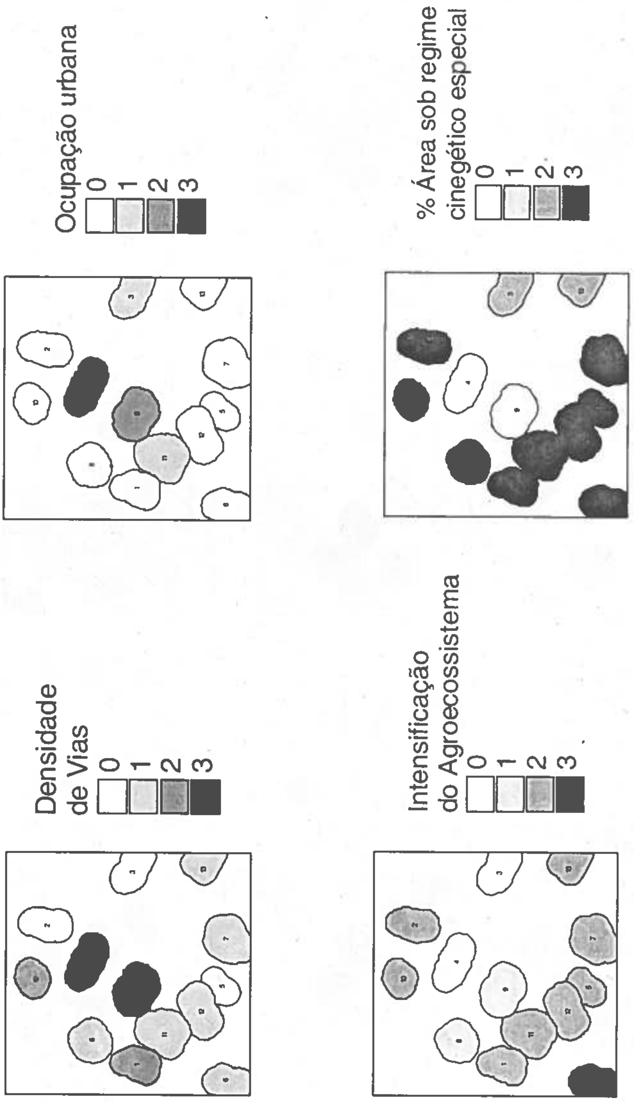

Figura 6 – Distribuição espacial dos factores de perturbação

Os índices de diversidade e heterogeneidade utilizados permitiram a identificação de áreas fragilizadas em termos de diversidade de *habitats* (7 e 13) e de áreas com estruturas semelhantes, como se verifica na zona envolvente da Ribeira de Valverde, na Zona a Sul e Sudeste de Évora, a Norte da cidade e na zona do Vale encaixado do Degebe. Verificou-se que as áreas com maior diversidade e heterogeneidade da paisagem se encontram na zona Sudoeste da AE (envolvente da Ribeira de Valverde) e no Vale encaixado do Rio Degebe, coincidindo com os solos de mais baixa qualidade agrológica e com os agroecossistemas tipo SPAE de sequeiro com produção de culturas forrageiras, bovinos e ovinos de carne, cereais e oleaginosas, classificados como agressivos (2) para o ecossistema. As zonas mais homogéneas e de menor diversidade de *habitats* são, a Sudeste, as áreas 7 e 13, e a Norte, a área 10, onde domina a cultura arvense de sequeiro em terra campa.

A análise da diversidade faunística relativamente à estrutura da paisagem permitiu verificar algumas tendências geográficas, nomeadamente entre a diversidade e heterogeneidade de *habitats* e a diversidade de aves. Observa-se uma coincidência espacial entre as áreas onde se regista uma maior diversidade de aves com as de maior diversidade de habitats (MacArthur & MacArthur, 1961).

Os anfíbios, quer adultos, quer formas larvares, não apresentam qualquer padrão de distribuição marcado. Verifica-se contudo alguma coincidência entre as áreas de maior diversidade de anfíbios adultos e as áreas de maior diversidade paisagística.

A diversidade de répteis é máxima na área 7, seguida pelas áreas 13, 8 e 11, mostrando uma tendência para as áreas de menor diversidade de habitats.

A análise da diversidade, disponibilidade de água e factores de perturbação permitiu observar que a diversidade de aves está fortemente relacionada com a disponibilidade de água, como demonstra a regressão linear determinada entre os dois factores (r^2=0.77) (figura 7). Este facto apoia os resultados empíricos verificados nos trabalhos de campo de Rabaça *et al.* (1995, 1996). Efectivamente a diversidade de aves é mais elevada ao longo da envolvente das linhas de água (principalmente da Ribeira de Valverde e do Rio Degebe), atingindo o máximo na área 6 – Albufeira da Tourega. As áreas com menor diversidade são as 7, 13 e 10, essencialmente compostas por culturas arvenses de sequeiro em terra campa.

A rede viária, embora cause perturbações ao nível da tranquilidade, não se poderá dissociar do factor ocupação humana, com o qual está fortemente relacionada, pelo que actuam em conjunto na perturbação da tranquilidade.

No que respeita aos agroecossistemas, verifica-se a coincidência entre a localização dos mais agressivos com a maior diversidade de *habitats* e de aves.

O tipo de exploração cinegética não é por si só, uma componente discriminadora de classes de diversidade faunística.

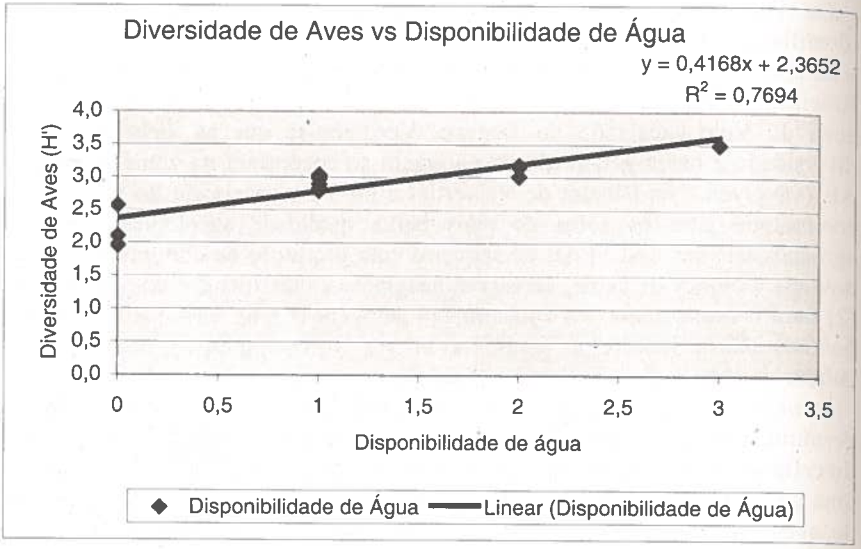

Figura 7 – Relação entre a diversidade de aves (H') e a disponibilidade de água nas áreas de pormenor. Regressão linear obtida por análise dos dados.

CONCLUSÕES

A presente análise embora evidencie já algumas relações espaciais entre os factores analisados, requer a realização de novos estudos, que permitam um maior detalhe e pressão de amostragem para que se confirme a sua utilidade para a monitorização da biodiversidade e gestão dos respectivos recursos e factores de perturbação.

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SADIND - SISTEMA DE VISUALIZAÇÃO INTERPRETATIVA PARA A GESTÃO AMBIENTAL DO ESTUÁRIO DO SADO

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A Reserva Natural do Estuário do Sado foi criada em 1980 com o intuito de fazer uma gestão racional do estuário do Sado. Com efeito, o estuário constitui um recurso local de elevada produtividade primária. A diversidade de flora e fauna existente nos inúmeros *habitats* do estuário, tornam-no num local de eleição para a reprodução de um elevado número de espécies.

Uma das maiores pressões sobre os meios naturais resulta da presença de um importante parque industrial, junto à fronteira da Reserva. Se bem que exista alguma informação sobre os impactes destas indústrias no sistema, encontra-se frequentemente dispersa ou em fontes diferenciadas tornando difícil a sua interpretação de conjunto. A necessidade de apurar mais em detalhe as relações causa-efeito com o intuito de apoiar a gestão ambiental da zona levou-nos a desenvolver o presente projecto.

Esta comunicação tem como objectivo apresentar o desenvolvimento de um sistema de visualização interpretativo para as unidades industriais a operar nas áreas envolventes do estuário do Sado e respectivos impactes. Primeiro fez-se uma caracterização destas unidades, identificando os tipos de poluição emitidos e os seus valores quantitativos. Posteriormente, registaram-se espacialmente recorrendo a Sistemas de Informação Geográfica (SIG), criando tabelas associadas com os descritores relevantes para os impactes gerados por estas unidades. O sistema aqui desenvolvido permite a realização de consultas à informação por parte do gestor ambiental e aceder em tempo real à visualização dos resultados.

Palavras chave: Estuário, Sado, Gestão Ambiental, SIG's.

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The Sado Estuary Nature Reserve was created in 1980 aiming at the rational management of the Sado estuary. In fact, the estuary is a local resource with a high primary productivity.
It is an elected site for a great number of species for reproduction due to the flora and fauna diversity in the various estuary habitats.
One of the greatest pressures on the natural system is due to the location of an important industrial site on the borderland of the RNES. Despite the availability of information about the impacts of these industries on the natural system, frequently the data is scattered or in various sources, making it difficult for the specialist to bring it together for an overall view. The need to assess in more detail the cause-effect relation to support environmental management, led us to develop the present project aiming to easy up the crossing and overlay of data from different sources.
The objective of this paper is to present the interpretative visualization of the industrial units operating in the surrounding areas of the Sado estuary. First, the team carried out a characterization of these units, identifying the various pollution emissions and their values. Afterwards, the industry units were registered spatially recurring to a GIS and creating associated tables with relevant descriptors for the impacts generated by these units.
SADIND, the visualization system developed here, allows the environmental manager the realization of queries and the visualized results in real time.

Keywords: Estuary, Sado, environmental management, GIS.

INTRODUÇÃO

Os estuários são considerados sistemas de grande importância no que respeita à conservação da natureza. Estes representam para a Estratégia Mundial da Conservação biótopos essenciais aos processos ecológicos que suportam a vida e portanto de vital importância para a manutenção dos equilíbrios ecológicos da biosfera (UICN, 1980 *in* APS, 1998). Estes são ainda considerados ecossistemas de elevada biodiversidade. Tudo isto, deve-se a um conjunto de características físicas e químicas que se encontram associadas a estas regiões, como é o caso do confinamento, do padrão de circulação das águas mais ou menos estabelecido pela influência das marés, da capacidade de armazenamento e reciclagem de nutrientes e energia (F. Ferreira, comunicação pessoal).

Segundo o Decreto-Lei nº 226/97 de 27 de Agosto (Directiva Habitats), os estuários integram a lista de habitats naturais e de interesse comunitário cuja conservação exige a designação de zonas especiais de conservação. Contudo, quer a sua importância, quer mesmo o estatuto que, de uma forma geral, estas regiões possuem ao nível da conservação da natureza, não são suficientes para as afastar das mais diversas pressões ambientais. O facto de se encontrarem entre o meio terrestre e marinho, torna-as não só frequentemente alvo de contaminações por poluentes, como também zonas reclamadas para os mais diversos usos por parte do Homem. Tudo isto acaba por contribuir para o aparecimento dos mais variados tipos de conflitos, o que torna difícil a gestão ambiental deste tipo de áreas.

De facto, da elevada disparidade de interesses e de responsabilidades que, de uma forma geral, governam estas regiões, têm resultado frequentemente situações de conflito que constituem por si só sérios obstáculos à intenção de desenvolver e implementar qualquer sistema de gestão ambiental.

Desta forma, a gestão costeira deve ser vista como um processo que governe esses conflitos de interesse e que distribua as actuais responsabilidades entre os actores-chave envolvidos. Ora, a área em que o presente estudo incide constitui precisamente um exemplo desse tipo de regiões. O estuário do Sado, localizado junto da península de Setúbal, para além de ser considerado uma das zonas húmidas mais importantes de Portugal, e portanto, de grande valor ambiental, comporta também um importante núcleo urbano, de elevado valor social e económico (APS, 1998). A este núcleo urbano encontra-se ligada uma intensa actividade económica, fundamentalmente associada à actividade industrial e portuária. A sua grande qualidade ambiental encoraja também a procura, criando enormes pressões. Tal deve-se em grande parte à localização estratégica desta região, pois encontra-se no centro de rotas marítimas, rodoviárias e aéreas, e simultaneamente, dispõe de bons acessos, quer para norte quer para sul, ou mesmo para a vizinha Espanha (COBA – HIDROTÉCNICA PORTUGUESA – WS ATKINGS – CONSULGAL – GIBB, 2000). A juntar a estes aspectos, convém não esquecer que esta região pertence à área metropolitana de Lisboa, sofrendo como tal a sua influência. A proximidade a este grande centro populacional contribui por si só para um maior desenvolvimento das actividades económicas que aqui se encontram instaladas. A actual restruturação da zona portuária da cidade de Lisboa, e a transferência de algumas das suas actividades vêm igualmente contribuir para a intensificação das já existentes no Porto de Setúbal, e consequentemente da actividade industrial (COBA – HIDROTÉCNICA PORTUGUESA – WS ATKINGS – CONSULGAL – GIBB, 2000).

As unidades industriais que se encontram em toda a área envolvente ao estuário, juntamente com a cidade de Setúbal, utilizam-no para descarga de efluentes e exercício das diversas actividades portuárias (CAEIRO *et al.*, 1999). A intensificação deste tipo de actividades juntamente com o próprio desenvolvimento que o Porto de Setúbal tem vindo a sofrer, tem contribuído para um incremento da pressão exercida, em particular sobre a área norte do estuário. Como consequência deste tipo de actividades, a qualidade da água, do sedimento e o próprio biota encontra-se actualmente bastante afectada, o que coloca em risco todo o seu equilíbrio ecológico (APS, 1998). Nas restantes áreas circundantes ao estuário, existem ainda actividades agrícolas, das quais se destacam as explorações de arroz, as salinas e um número crescente de aquaculturas. Dentro do próprio estuário, são ainda conduzidas intensas actividades de exploração de recursos piscícolas, conquícolas e isco.

Desta forma, torna-se urgente o desenvolvimento de ferramentas eficazes para que se possa proceder a uma correcta gestão ambiental de toda esta área natural.

Para o desenvolvimento de qualquer estratégia de Gestão Ambiental, será conveniente proceder-se previamente à sistematização de toda a informação, de forma a que a mesma possa servir de base para o desenvolvimento e implementação de um plano estratégico de acção. Tal sistematização deverá incluir, por exemplo, a identificação e hierarquização de todos os valores naturais, de todas as fontes poluidoras, e ainda dos usos potenciais que os mesmos valores possam vir a ter. Só dessa forma será possível o desenvolvimento de acções que, por um lado estejam direccionadas à protecção e preservação dos principais valores naturais, e por outro à reabilitação e recuperação de zonas contaminadas e/ou ambientalmente degradadas.

O presente trabalho resulta precisamente não só da necessidade de sistematização e gestão eficaz de toda a informação, em particular da relacionada com as fontes de poluição existentes nas áreas envolventes ao Estuário do Sado, como também de fazer com que a mesma se encontre organizada por forma a que facilmente permita o seu cruzamento com outro tipo de informação. Para uma gestão efectiva é de grande importância que a informação seja georeferenciada.

Este trabalho tem assim como objectivo o desenvolvimento de um sistema de visualização interpretativo para as unidades industriais em operação nas áreas envolventes ao estuário. As vantagens acrescidas deste tipo de tratamento da informação, que nos permite cruzar dados descritivos com georeferenciação espacial são consideráveis para desenvolver estratégias de gestão e têm sido pouco exploradas de uma forma sistemática. Este sistema integra dois tipos de informação: por um lado, a espacial, através da utilização de um Sistema de Informação Geográfica e, por outro, dados da própria fonte poluidora, estruturada numa base de dados. Isto permitirá a geração de mais informação, na medida em que a integração utilizada foi estruturada para possibilitar uma maior facilidade e eficácia no cruzamento da informação armazenada com dados de outro tipo, nomeadamente com outras imagens georeferenciadas onde, por exemplo, seja possível observar o estado de qualidade da água.

METODOLOGIA

Recolha de Informação

Numa primeira fase, procedeu-se ao levantamento de informação das principais unidades industriais a operar nas áreas envolventes do estuário do Sado. A informação recolhida resultou, fundamentalmente, da compilação de dados existentes em três trabalhos, nomeadamente: o Plano Municipal de Ambiente de Setúbal (PMAS), o Plano de Ordenamento e Expansão do Porto de Setúbal (POEPS) e o Plano de Bacia Hidrográfica do rio Sado (PBHS).

No primeiro destes três documentos - PMAS - é possível encontrar um conjunto de "fichas ambientais" para cada uma das principais actividades económicas em_funcionamento no Concelho de Setúbal, na margem direita do Estuário do Sado. Estas fichas foram elaboradas com o objectivo de sistematizar e registar informação relativa às potenciais emissões e respectivos sistemas de tratamento, e ainda uma primeira estimativa do impacte ambiental decorrente de cada actividade económica nos domínios ambientais ar, água e solo.

O POEPS contém o mesmo tipo de informação com um menor grau de pormenor, apresentando, no entanto, informação relativa a algumas unidades industriais não referidas no PMAS.

Por último, no PHBS é possível encontrar uma lista exaustiva de actividades económicas a operar em toda a área da Bacia Hidrográfica do rio Sado e ainda informação genérica destas actividades.

Os critérios adoptados para a selecção das unidades industriais a incluir na base de dados basearam-se na sua dimensão, potencial impacte e existência de informação ambiental detalhada. O documento que se revelou mais adequado para esta selecção foi o PMAS.

Construção da Base de Dados

Finalizada a compilação de toda a informação procedeu-se à construção de uma base de dados. A necessidade desta advém fundamentalmente de três situações: permitir para cada unidade industrial a agregação de informação relativa a diversos domínios, bem como a facilidade de gestão e acesso à mesma; gerir e cruzar facilmente um grande volume de dados; permitir também que a informação se encontre disponível sob a forma de tabelas relacionadas para assim servir de apoio ao sistema de visualização interpretativo.

Foram assim construídas quatro tabelas (Figura 1): a primeira com o nome de "Tabela Geral" onde se encontra a designação e localização de todas as unidades industriais consideradas, e ainda informação genérica sobre a actividade económica propriamente dita; as restantes tabelas são relativas a cada um dos descritores ambientais considerados (Ar, Água e Solo).

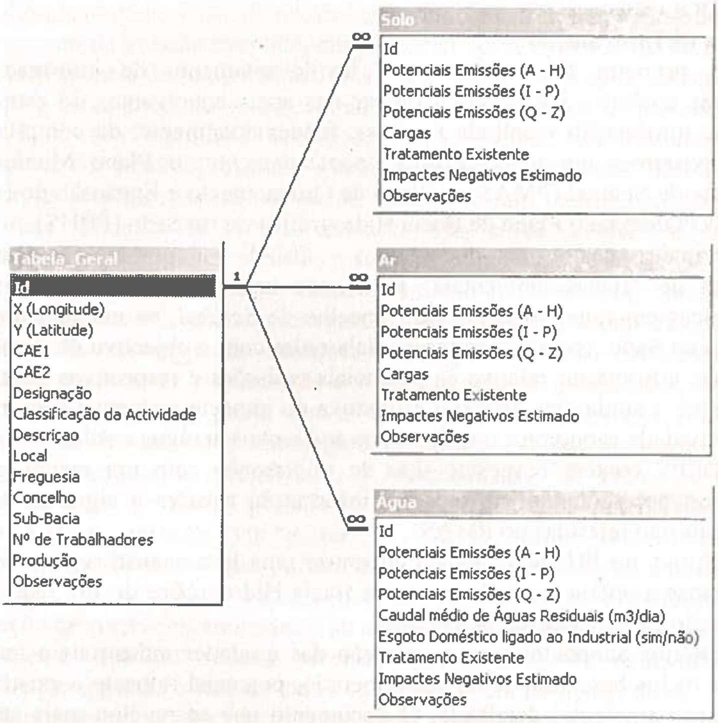

Figura 1 – Esquema das relações entre as tabelas construídas e de todos os seus atributos.

Dada a inexistência de informação quantitativa para a maioria dos casos considerados, apenas se procedeu a uma sistematização de informação de carácter mais qualitativo.

Construção do Sistema de Visualização Interpretativo

Tal como referido, o Sistema de Visualização Interpretativo foi construído recorrendo a Sistemas de Informação Geográfica. O Programa utilizado foi o "*ArcView 3.2* ®".

O registo espacial de todas as unidades industriais consideradas neste estudo foi feito tendo por base um mapa em formato digital e georeferenciado, disponibilizado pela Câmara Municipal de Setúbal (CMS), com a localização de todas as infra-estruturas existentes no concelho. Não obstante, para um registo mais preciso pensa-se numa fase posterior recorrer a instrumentos que se baseiem em sistemas de posicionamento global (GPS).

Uma vez identificadas foram elaborados a partir do mapa base dois tipos de coberturas: de pontos e de polígonos (Figs. 2 e 3). O primeiro serviu para a identificação espacialmente das várias unidades industriais e o segundo para contabilizar e visualizar as respectivas áreas ocupadas.

No "*ArcView*" sempre que é Estas tabelas podem posteriormente ser cruzadas com dados registados em "*Microsoft Access*" (programa de construção de base de dados) de modo a tirar partido das operacionalidades oferecidas.construída umEsta ligação irá permitir que o "*ArcView*" possa encontrar e usar todas as tabelas e consultas ou apenas as seleccionadas, já previamente construídas na Base de Dados.a cobertura é gerada automaticamente uma tabela com o registo de todos os pontos/polígonos feitos.

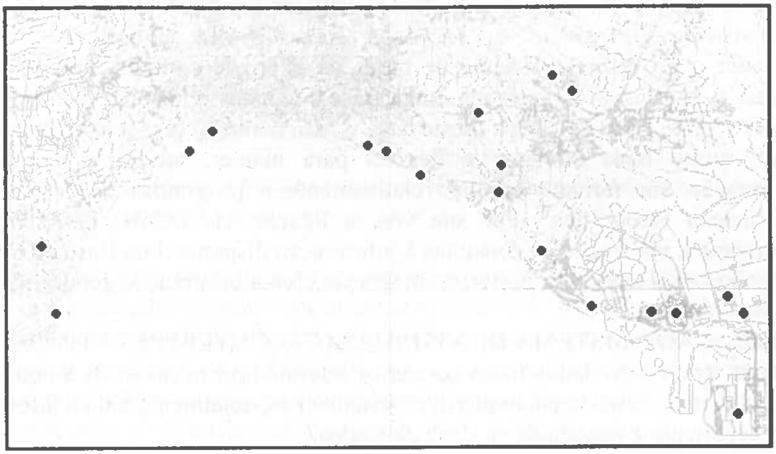

Figura 2 – Mapa base com cobertura de pontos.

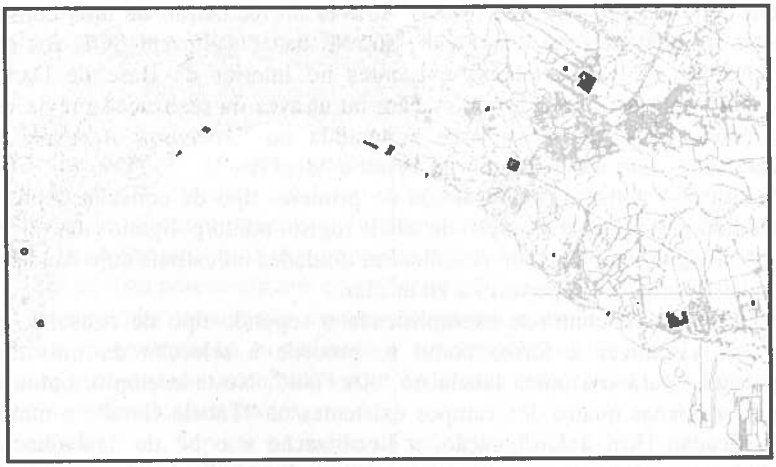

Figura 3 – Mapa base com cobertura de polígonos.

Procedeu-se assim à ligação via "*Open DataBase Connectivity*" (ODBC) da informação espacial de cada unidade industrial gerada pelo"*ArcView*" e a restante informação que já se encontra organizada e registada na Base de Dados.

Esta ligação é dinâmica e não estática, isto é, qualquer modificação feita na Base de Dados será imediatamente transmitida sempre que o documento for aberto ou sempre que o operador desejar actualizar a informação existente no "*ArcView*".

Ainda antes de finalizar é importante justificar o porquê da integração destes dois sistemas ("*ArcView*" e "*Microsoft Access*"). Ora, embora o "*ArcView*" possuam o seu próprio sistema de bases de dados, é contudo mais vantajoso manter a informação organizada numa Base de Dados relacional. Um programa especifico, de Base de Dados (neste caso o "*Microsoft Access*") é, de uma forma geral, muito mais eficiente e flexível para manter, modificar e consultar informação em formato tabular relativamente a programas de Sistemas de Informação Geográfica[1]. Por sua vez, a ligação via ODBC permite muito facilmente a realização de consultas à informação disponível na Base de Dados a partir do "*ArcView*" para posterior integração com a informação geográfica.

APLICAÇÃO – SISTEMA DE VISUALIZAÇÃO INTERPRETATIVO

Após terem sido dados todos os passos intermédios, necessários à construção deste sistema, torna-se então possível visualizar espacialmente toda a informação já previamente armazenada na Base de Dados.

Contudo, para que tal seja possível é necessária a realização prévia de consultas, isto é, a selecção prévia da informação que se deseja visualizar. Esta tarefa pode ser feita de três formas: através da realização de uma consulta às tabelas já existentes no "*ArcView*"; através da consulta em SQL aos campos pertencentes às várias tabelas existentes no interior da Base de Dados, no momento em que é feita a importação; ou através da realização prévia de uma selecção/consulta da informação pretendida no "*Microsoft Access*", a qual poderá ser posteriormente importada para o "*ArcView*".

A Figura 4 constitui um exemplo do primeiro tipo de consulta. Após se ter procedido previamente à junção da tabela registo ponto/polígono com os registos da tabela água, optou-se por visualizar as unidades industriais cujo caudal médio de águas residuais é superiores a 70 m^3/dia.

Na Figura 5 encontra-se exemplificada o segundo tipo de consulta. Aqui é possível visualizar a forma como se procede à selecção da informação e importação para um única tabela no "*ArcView*". Neste exemplo, optou-se por importar apenas quatro dos campos existentes na "Tabela Geral": o número de identificação (Id), a Designação, a Localização e o Nº de Trabalhadores, e apenas os registos que apresentam um número de trabalhadores superior a 150.

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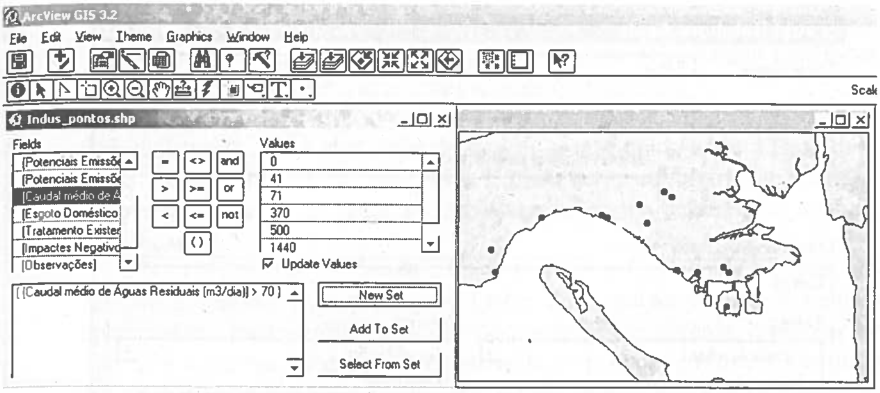
[1] http://www.ar.utexas.edu/Courses/parmenter/gis/tips/access.html (Agosto 2001).



([Caudal médio de Águas Residuais (m3/dia)] > 70)

Figura 4 – Visualização das unidades industriais (pontos mais claros) com valores de caudal médio de águas residuais superiores a 70 m^3/dia.

Os dois tipos de consultas anteriores apresentam como vantagem o facto de permitirem que o operador trabalhe unicamente em "*ArcView*" para a sua realização.

Contrariamente, na terceira forma de consulta o operador é obrigado a utilizar simultaneamente os dois programas (*ArcView* e *Access*). Contudo, esta mostra-se nalguns aspectos mais vantajosa: elevada simplicidade, flexibilidade e rapidez no acesso, selecção e obtenção da mais variada informação. Permite consultar informação existente em diversos campos pertencentes a diferentes tabelas, seleccionar informação de acordo com uma grande variedade de critérios, e simultaneamente manter a consistência da mesma já que no desenrolar destas acções são satisfeitas as várias relações previamente definidas entre as diferentes tabelas.

CONCLUSÃO

De acordo com o descrito anteriormente, conclui-se que o sistema desenvolvido neste trabalho poderá constituir uma ferramenta eficaz e complementar ao desenvolvimento e implementação de um sistema de gestão ambiental para o Estuário do Sado, em particular para as áreas protegidas que o mesmo alberga. Esta potencialidade está relacionada com diversos aspectos:

- por proporcionar uma maior rapidez no acesso e visualização de grandes volumes de informação, actualizada e em tempo real, de facto, o sistema permite que os dados recolhidos num dado momento possam ser imediatamente visualizados/analisados;
- por constituir um suporte à criação de múltiplas formas de visualização da mesma;
- pela possibilidade de aplicação a outros tipos de informação, como por exemplo, a outro tipo de actividades económicas igualmente consideradas como fontes poluidoras (explorações agrícolas, pecuárias, etc.);

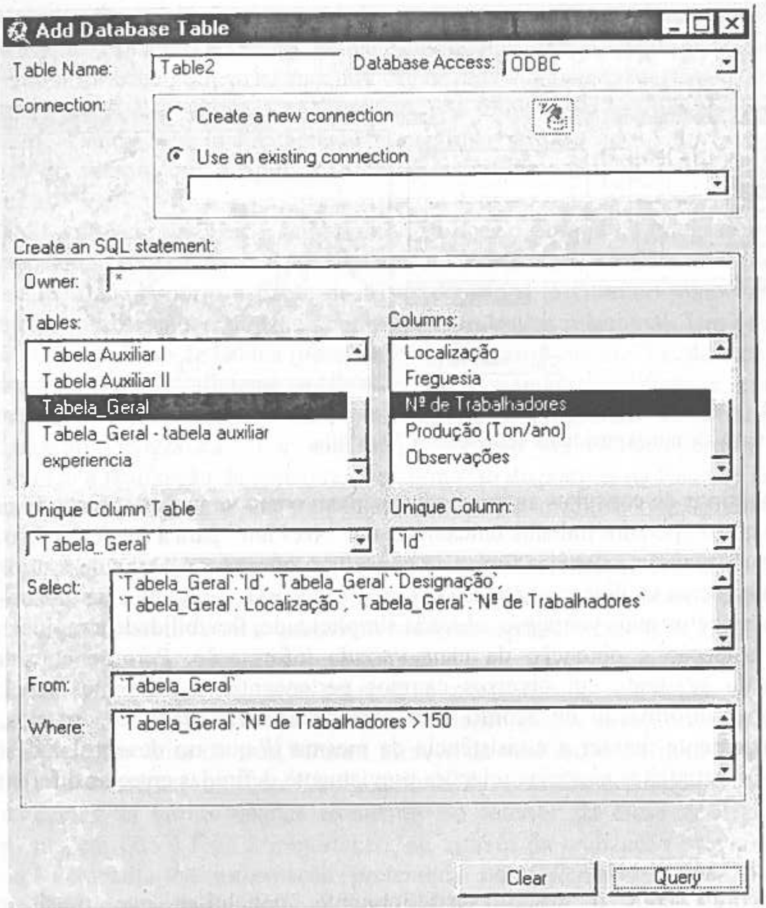


Table Name: Table2
Database Access: ODBC

Connection:
○ Create a new connection
● Use an existing connection

Create an SQL statement:
Owner: *

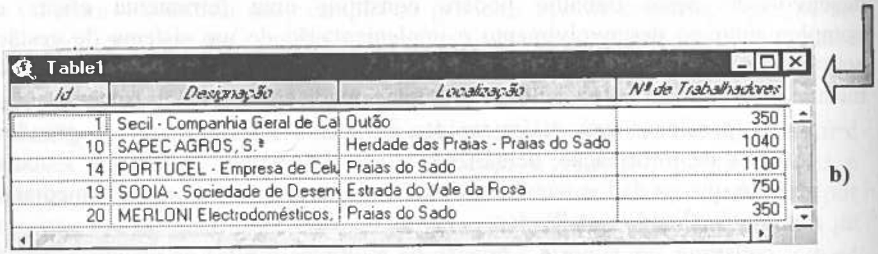
Tables:
- Tabela Auxiliar I
- Tabela Auxiliar II
- Tabela_Geral
- Tabela_Geral - tabela auxiliar
- experiencia

Columns:
- Localização
- Freguesia
- Nº de Trabalhadores
- Produção (Ton/ano)
- Observações

Unique Column Table: `Tabela_Geral`
Unique Column: `Id`

```
Select: `Tabela_Geral`.`Id`, `Tabela_Geral`.`Designação`,
        `Tabela_Geral`.`Localização`, `Tabela_Geral`.`Nº de Trabalhadores`
From:   `Tabela_Geral`
Where:  `Tabela_Geral`.`Nº de Trabalhadores`>150
```

a)



Id	Designação	Localização	Nº de Trabalhadores
1	Secil - Companhia Geral de Cal	Outão	350
10	SAPEC AGROS, S.ª	Herdade das Praias - Praias do Sado	1040
14	PORTUCEL - Empresa de Celu	Praias do Sado	1100
19	SODIA - Sociedade de Desenv	Estrada do Vale da Rosa	750
20	MERLONI Electrodomésticos,	Praias do Sado	350

b)

Figura 5 – a) Exemplo de consulta em SQL, que permite a selecção de quatro campos e dos registos cujo número de trabalhadores correspondente é superior a 150; b) Tabela resultante.

- por fim, por constituir um primeiro passo para a sistematização, organização, gestão e geração de informação relacionada com os diversos domínios ambientais existentes no Estuário do Sado.

É ainda de referir que o método utilizado para a construção deste sistema, nomeadamente ao nível do tipo de ligação estabelecida entre a Base de Dados e o Sistema de Informação Geográfica, pode ser particularmente vantajoso para situações, onde a informação armazenada é frequentemente actualizada e modificada.

Embora a informação recolhida, armazenada e organizada na Base de Dados relativa a algumas das principais fontes de poluição industrial tenha servido apenas como exemplo para o desenvolvimento deste sistema, espera-se no futuro não só a continuação dessa sistematização como também que a mesma venha a ser cruzada com dados de níveis de poluição já recolhidos para o estuário, como contributo base para a gestão ambiental do estuário.

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A FRAGMENTAÇÃO COMO CAUSA PRINCIPAL DA REDUÇÃO DO HABITAT DE *COREMA ALBUM* NA SUA ÁREA DE DISTRIBUIÇÃO

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Clavijo, A., Díaz Barradas, M. C., Ain-Lhout, F., Zunzunegui, M. & Correia, O. (2002). A fragmentação como causa principal da redução do habitat de *Corema album* na sua área de distribuição. ***Revista Biol. (Lisboa)*** **20**: 109-120.

A camarinha (*Corema album*) é uma espécie dióica, endémica das dunas atlânticas da Península Ibérica, cuja área de distribuição se estende desde Finisterra no NW da Galiza, até às proximidades de Gibraltar como limite meridional.
Os objectivos deste trabalho consistiram em identificar e caracterizar as populações da camarinha ao longo de toda a sua área de distribuição, registando a sua localização geomorfológica (1° e 2° vale interdunar, dunas interiores, dunas sobre arribas), a extensão das populações, as espécies vegetais companheiras, a pressão humana, a competição com outras espécies, o tamanho médio dos indivíduos e a existência de regeneração natural.
Os resultados deste estudo mostraram que só existem duas zonas com populações extensas de *C. album* apresentando indivíduos de todas as classes de idade, uma no litoral português entre Nazaré e Ovar, e a outra na costa de Huelva (Espanha) no sistema de dunas do Asperillo (Parque Natural de Doñana), existindo outra população importante nas dunas de Tróia. As outras populações estão limitadas a zonas de areia sobre arribas, como as populações da costa vicentina, ou estão reduzidas a fragmentos, destacando-se as populações da Galiza, da foz do Minho, do Algarve, ou as de Bolonia na província de Cádiz. Estas populações fragmentadas, reduzidas a ilhas de areia sem regeneração natural, estão formadas, na sua maioria, por indivíduos velhos, e a sua

sobrevivência no futuro depende de medidas de protecção e regeneração de habitats.

Palavras chave: Dunas, litoral atlântico, índice de impacto, ilhas de areia.

Clavijo, A., Díaz Barradas, M. C., Ain-Lhout, F., Zunzunegui, M. & Correia, O. (2002). The habitat fragmentation as the main cause of habitat reduction in *Corema album* along its distribution area. *Revista Biol. (Lisboa)* **20**: 109-120.

Corema album is a dioecious species, endemic of the Atlantic dunes of the Iberian Peninsula, extending along the coast from Finisterre in the NW of Galicia to Gibraltar.

The objectives of this study were to identify and characterize the populations of *C. album* along its biogeographical area, recording its geomorphological location (foredunes, slacks, inner dunes or dunes over cliffs), the size of the populations, the plant community composition, the level of human pressure, the role of competition by other species, the average size of the individuals and the existence of natural regeneration.

The results showed that only two large populations of *C. album* still remain, presenting individuals of all age classes, one located in the Portuguese coast from Nazaré to Ovar, and the other in the Huelva coast (Spain) in the dune system of Asperillo (Doñana Natural Park). There is also an important population at the Tróia dunes. The other populations are limited to coastal cliffs, such as in the Vicentin coast, or are fragmented, like the populations of Galicia, Minho estuary, Algarve or Bolonia in Cadiz region. These fragmented populations, limited to islands, without natural regeneration, are mainly composed of old and large individuals and their survival depends on land protection and habitat regeneration.

Keywords: Dunes, Atlantic coast, impact index, sand islands.

INTRODUÇÃO

Actualmente, considera-se que a fragmentação e a degradação dos habitats são das principais causas da extinção local e regressão de numerosas espécies (FAHRIG, 1997, CARLSON, 2000, CASTILLO, 2001).

Os ecossistemas de dunas litorais da Península Ibérica têm sofrido uma enorme regressão nos últimos 30 anos devido à enorme pressão para construir urbanizações, campos de golf, estradas, e à introdução de espécies exóticas de crescimento rápido com o fim de fertilizar o sistema geralmente pobre em nutrientes e fixar o substrato. Estes processos têm posto em perigo a sobrevivência ou a regeneração de numerosas espécies autóctones, como por exemplo o cedro das dunas (*Juniperus oxycedrus*) e a camarinha (*Corema album*).

À medida que se reduz o seu território, a área das espécies autóctones, geralmente com taxas de crescimento lento, fica reduzida a fragmentos que em algumas ocasiões podem manter certo fluxo genético entre eles, sendo considerados como metapopulações (LEVINS, 1969, 1970), mas em outros casos aparecem isolados, como ilhas, com escasso número de indivíduos, submetidas a processos de deriva genética e extinção local das populações.

A camarinha, *Corema album* (L.) D. Don (*Empetraceae*) é um arbusto dióico, com flores pequenas e polinização anemófila que pode atingir um metro de altura. O número de flores por inflorescência varia conforme os sexos mas nunca excede as 20 flores (GUITIÁN *et al.,* 1997). O fruto é uma baga mais ou menos esférica (5-8 mm de diâmetro), branca ou rosada. A espécie está descrita como estritamente dióica (VALDÉS *et al.,* 1987) embora se tenham encontrado alguns exemplos de androdioicia na costa de Huelva.

Esta espécie é endémica da costa atlântica da Península Ibérica, ocupando areias mais ou menos móveis em sistemas dunares ou em areias sobre arribas rochosas, habitats que têm sofrido um crescente impacto humano nos últimos tempos. Recentemente, encontrou-se uma população de *C. album* na costa mediterrânica espanhola e também existe uma subespécie nos Açores (*C. album* subsp. *azoricum* Pinto da Silva) que coloniza lavas e cinzas vulcânicas.

Dada a enorme importância desta espécie nos sistemas dunares do ocidente peninsular e a crescente regressão destas populações (DIAZ BARRADAS *et al.*, 2000) foi objectivo deste trabalho identificar e caracterizar as populações da camarinha em toda a sua área de distribuição, que se estende ao longo de um gradiente climático desde clima atlântico no norte, a clima mediterrânico no sul. Neste sentido, foram desenvolvidos os seguintes objectivos específicos: (i) identificar a localização geomorfológica das populações (1° e 2° vale interdunar, dunas interiores, dunas sobre arribas), (ii) quantificar a extensão das mesmas (densidade e cobertura) e o tamanho dos indivíduos, (iii) registar as espécies vegetais companheiras e a pressão humana, (iv) e a existência de regeneração natural.

MATERIAL E MÉTODOS

Áreas de estudo

Ao longo da área de distribuição de *Corema album* escolheram-se 11 populações diferentes, representativas de todo o gradiente desta espécie, desde a zona termomediterrânea no sul de Espanha e Portugal, até à zona mesomediterrânea no Norte da Península.

As zonas escolhidas, com o correspondente diagrama ombrotérmico, encontram-se representadas na Figura 1 e foram as seguintes:

- Vila de Camariñas, situada no noroeste da Galiza, corresponde à população mais setentrional desta espécie.

- Dunas situadas na foz do Minho, próximo de Caminha.
- Dunas de São Jacinto: situadas na Reserva Natural do mesmo nome.
- Dunas litorais situadas próximo da vila de Mira.
- Populações no Cabo Carvoeiro (Peniche).
- Dunas da Fonte da Telha, situadas na Paisagem Protegida da Arriba Fóssil da Costa da Caparica.
- Dunas de Tróia (Reserva Natural do Estuário do Sado).
- Populações situadas junto da praia de Monte Clérigo, situadas na Reserva Natural do Sudoeste Alentejano e Costa Vicentina.
- Mata de Monte Gordo.
- Dunas do Asperillo, no Parque Natural de Doñana (Huelva).
- Punta Camarinal, próximo da Praia de Bolonia (Cádiz).

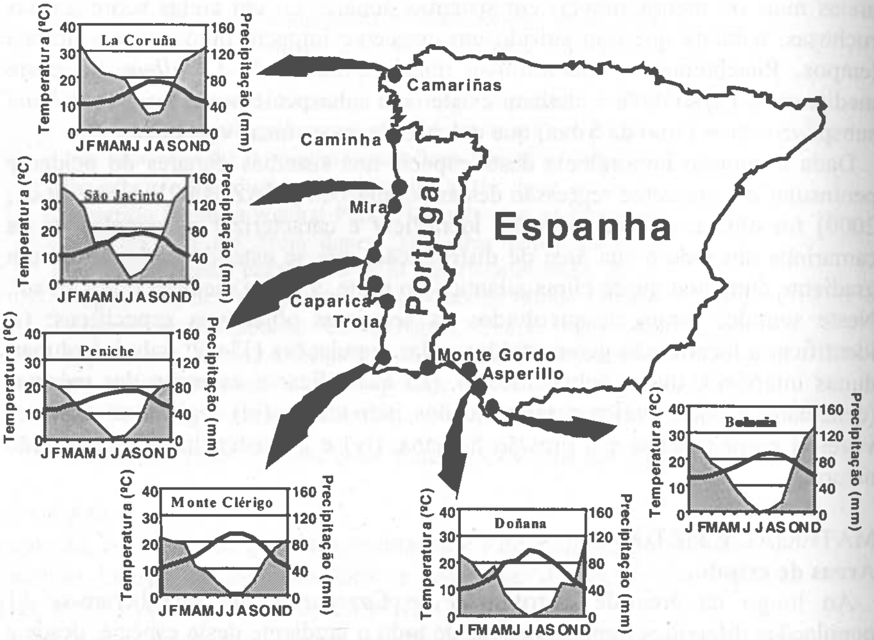

Figura 1. - Localização das áreas de estudo na Península Ibérica ao longo do gradiente climático e representação do diagrama ombrotérmico de Gaussen para algumas das localidades em estudo.

Avaliação das populações

Para avaliar o estado das populações em cada uma destas zonas de estudo, realizou-se um transecto paralelo à linha de costa, de 200 m de comprimento e 10 m de largura onde foram estudadas as seguintes variáveis:

Localização geomorfológica

Para uma melhor caracterização das zonas de estudo, agruparam-se as populações em três tipos atendendo à sua localização geomorfológica:

- Populações sobre ecossistemas tipicamente dunares, situadas atrás da primeira duna que podem manter-se em estádios sucessionais mais avançados. É o caso do Asperillo, Mira, São Jacinto, Fonte da Telha e Tróia.
- Populações em areias sobre arribas, onde a intensidade do vento pode ser o factor ecológico principal, modelando tapetes de camarinha, que não superam os 30 cm de altura. Nesta situação encontram-se as populações do Cabo Carvoeiro e Monte Clérigo.
- Populações afectadas por repovoamentos ou pela pressão humana. Ao longo de toda a sua área de distribuição pode-se observar que a camarinha resiste muito mal à competição com espécies de crescimento mais rápido, como os pinheiros (*Pinus* sp.), a *Acacia* sp. e o *Carpobrotus edulis*. Os exemplares estudados aparecem em muitos casos afogados, ou restringidos a "*gaps*", com uma morfologia muito aberta e um índice de área foliar (LAI, "*leaf area index*") muito baixo. Tal é o caso das populações de Bolonia, Monte Gordo e Caminha.

Densidade das populações e tamanho dos indivíduos

Em 10 parcelas de 10 x 10 m, separadas 10 m entre si, foram realizadas medidas populacionais e morfológicas de *C. album*. Em cada parcela, contou-se o número de copas em indivíduos reprodutores e não reprodutores, segundo a definição de "*ramets*"de KAYS & HARPER (1974). Mediram-se os respectivos diâmetros, maior e menor, da copa. A cobertura foi determinada a partir da projecção da copa calculada a partir do diâmetro maior e menor, considerando a superfície de uma elipse.

Para estimar a densidade das populações, tomou-se o número de copas por metro quadrado, em cada transecto estudado.

Regeneração natural

Para avaliar a regeneração das populações das diferentes zonas, contabilizou-se e mediu-se a cobertura de todos os indivíduos não reprodutores (ou aparentemente não reprodutores) nas diferentes parcelas do transecto. Os indivíduos foram agrupados em cinco classes de tamanho, representativas das diferentes classes de idade.

As classes de tamanho consideradas foram as seguintes: menor ou igual a 1 cm^2, entre 1 e 10 cm^2, entre 10 e 100 cm^2, entre 100 e 1000 cm^2 e maior de 1000 cm^2. Os indivíduos de mais de 1000 cm^2 podem dar lugar a confusões, pois podem ser adultos sem flor.

Espécies acompanhantes

Em cada transecto, mediu-se a cobertura de todas as espécies, incluindo *Corema album*, utilizando o método dos quadrados pontuais, onde em pontos distanciados de 25 cm foi anotada a presença das diferentes espécies. Foi igualmente calculado o Índice de diversidade de Shannon (SHANNON & WEAVER, 1949) para cada um dos transectos.

As zonas foram agrupadas em três tipos conforme o tipo de vegetação dominante:

- Matas com pinheiros.
- Matas sobre dunas.
- Zonas degradadas.

Pressão humana

Para avaliar a pressão humana consideraram-se três parâmetros:

- Existência de algum tipo de protecção para a zona.
- Nível de povoamento humano.
- Tipo de uso da zona (turístico, caça, natural sem grandes intervenções).

A cada zona foram atribuídos índices para cada um destes parâmetros de acordo com o grau de intervenção (tabela 1).

A classificação utilizada atribuiu uma pontuação elevada às acções humanas pouco conservadoras e pontuações baixas àquelas acções pouco intervencionistas com o meio natural. Para cada zona, os pontos foram adicionados, obtendo-se uma maior pontuação para um maior grau de pressão humana. O nível máximo de pressão humana obtido foi de 5.

Tabela 1. Nível de pressão humana

Figura da protecção		**População**		**Uso da zona**	
Característica	Índice	Característica	Índice	Característica	Índice
Sim	0	Alto	2	Natural	0
Não	1	Médio	1	Caça ou similar	1
		Baixo	0	Turismo	2

Critérios de avaliação

Os critérios para avaliação das populações encontram-se resumidos na tabela 2. Para estabelecer o índice numérico, seguiu-se o mesmo critério que para a análise da pressão humana. Um índice numérico baixo, corresponde a acções ou situações que implicam um bom estado populacional, enquanto um índice elevado corresponde a acções ou situações que implicam ou conduzem a uma degradação da população. Desta forma, uma população em excelentes condições seria aquela que estivesse situada num ambiente tipicamente dunar, com uma densidade populacional maior a 0,4 copas/m^2, com um tamanho médio de indivíduo entre 0 e 1 m, com uma população de não reprodutores representada por todas as classes de idade e com uma pressão humana baixa (zona protegida, pouco povoada e não alterada).

Tabela 2. Critérios de avaliação.

Variável	Característica	Classificação	Índice numérico
LOCALIZAÇÃO GEOMORFOLÓGICA	Ecossistemas sobre dunas	Duna	0
	Populações sobre arribas	Arriba	1
DENSIDADE DA POPULAÇÃO	Repovoamento ou pressão humana	Repovoamento	2
	Entre 0 e 0,2 copas/m^2	Baixa	2
	Entre 0,2 e 0,4 copas/m^2	Média	1
	Maior de 0,4 copas/m^2	Alta	0
TAMANHO DOS INDIVÍDUOS	Entre 0 e 1 m^2	Pequeno	0
	Entre 1 e 3 m^2	Médio	1
	Maior de 3 m^2	Grande	2
FLORA COMPANHEIRA	Vegetação dunar	Dunas	0
	Mata com pinheiros	Pinheiro	1
	Zonas degradadas	Degradado	2
REGENERAÇÃO NATURAL	5 classes de idade	Alta	0
	50% de 10 a 100 cm^2	Média	1
	50% de 100 a 1000 cm^2 ou mais de 1000 cm^2	Baixa	2
PRESSÃO HUMANA	Entre 0 e 2 pontos	Baixa	0
	Entre 3 e 4 pontos	Média	1
	5 pontos	Alta	2

RESULTADOS E DISCUSSÃO

O tamanho médio dos indivíduos e a densidade das populações estão representados na Figura 2. Pode-se apreciar que existem grandes diferenças entre elas, destacando Caminha com os indivíduos de maior porte (5,5 m^2 de projecção individual) e uma densidade muito baixa (<0,1 copas/m^2), em oposição a São Jacinto e Camariñas onde o tamanho médio dos indivíduos oscila à volta de 0,5 m^2, mas a densidade populacional é elevada (>0,4 copas/m^2). Estes resultados parecem indicar que as populações de Caminha estão formadas por indivíduos velhos, com uma grande cobertura individual, em oposição a São Jacinto e Camariñas que parecem ser populações em expansão. As populações sobre arribas (Peniche e Monte Clérigo) apresentam valores de densidade média e tamanhos relativamente pequenos. A principal diferença com as outras populações está na altura dos indivíduos que apresentam formas rasteiras devido aos fortes ventos (DIAZ BARRADAS *et al.*, 2000).

Na Figura 3, representa-se o Índice de diversidade de Shannon nos diferentes locais estudados, bem como a percentagem de cobertura total de *Corema album*. Aparentemente, parece existir uma relação negativa entre estes dois parâmetros, onde uma maior cobertura desta espécie parece estar relacionada com uma menor diversidade (casos de Bolonia e Mira). Contudo, não se verificou uma correlação significativa entre estes dois parâmetros (coeficiente de correlação de Pearson). A maior cobertura da camarinha encontrou-se no Asperillo e a menor em Monte Gordo.

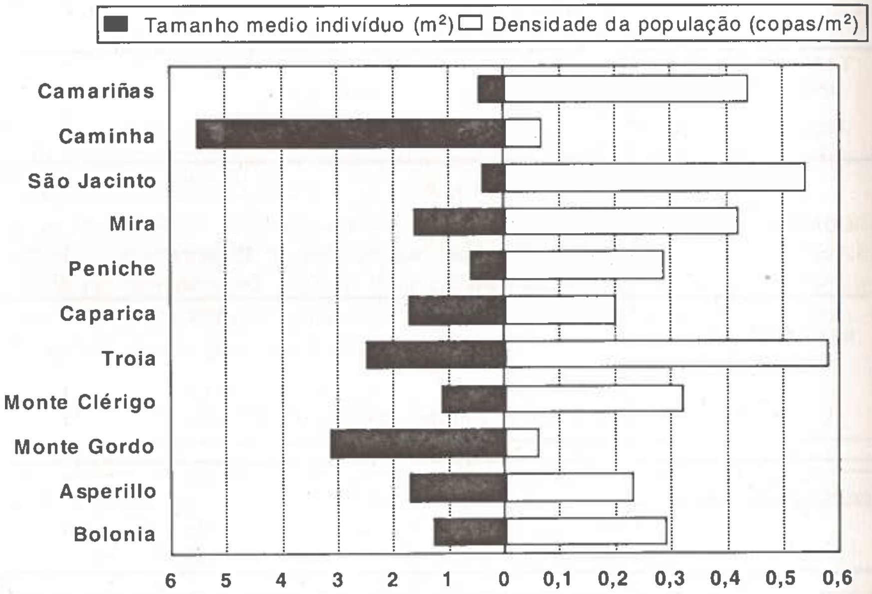

Figura 2. Densidade das populações (nº de copas/m^2) e cobertura média dos indivíduos (m^2) nas diferentes populações estudadas.

A regeneração natural, medida como a existência de não reprodutores das diferentes classes de tamanho, apresentou grandes diferenças nas populações estudadas (Fig. 4). Monte Gordo não aparece na figura, pois não apresenta indivíduos não reprodutores. As populações de Mira, São Jacinto e Asperillo,

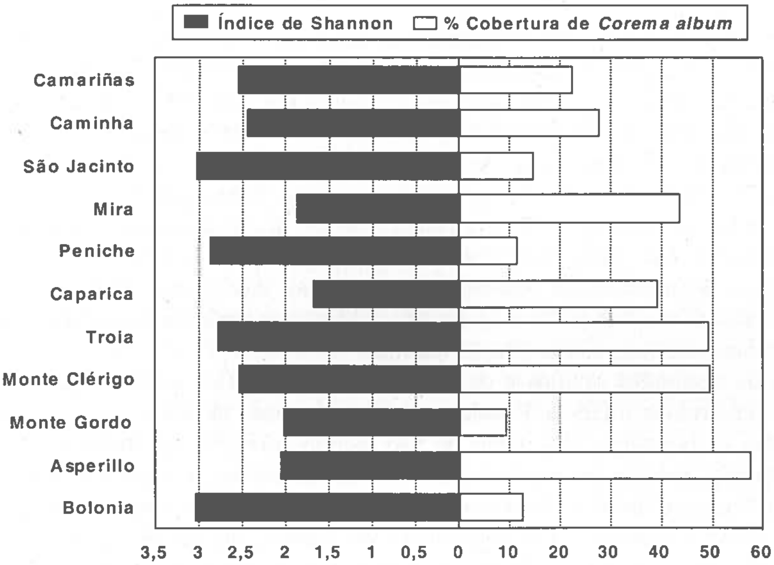

Figura 3. Comparação do índice de diversidade de Shannon e percentagem de cobertura total de *Corema album* nas diferentes populações estudadas.

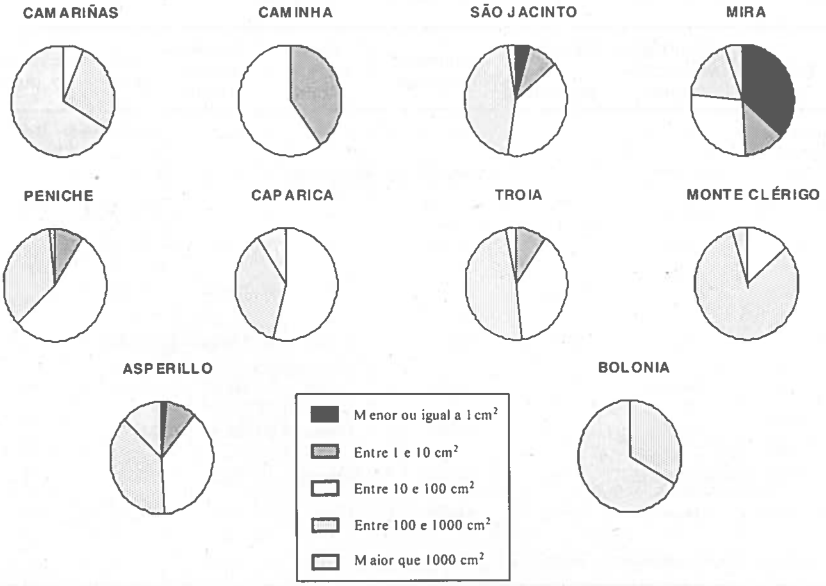

Figura 4. Distribuição dos indivíduos não reprodutores por diferentes classes de tamanho.

apresentam indivíduos não reprodutores distribuídos por todas as classes de tamanho ainda que em diferentes proporções, ao contrário de Bolonia e Caminha, que apresentam uma distribuição apenas por duas classes de tamanho.

A reprodução de *C. album* parece estar associada a vectores animais. As bagas desta espécie são consumidas por diferentes vertebrados, coelhos, raposas, gaivotas e melros (CALVIÑO, 2000), sendo a germinação das sementes facilitada pela passagem pelo aparelho digestivo destes animais (ÁLVAREZ, sem publicar). Nos sítios mais perturbados, a ausência de animais torna quase impossível a regeneração natural, ao contrário nas zonas protegidas ou conservadas como Asperillo, São Jacinto e Mira, a existência de comunidades preservadas permitem a regeneração natural.

Com os resultados obtidos e de acordo com os critérios estabelecidos neste artigo, elaborou-se a tabela 3 onde se resume o estado de todas as populações estudadas. A população das dunas de São Jacinto apresenta um índice 0, o que revela uma população em bom estado e em total expansão, pois além de ser uma zona protegida, trata-se de dunas de recente deposição. Os maiores valores deste índice foram observados nas populações de Caminha e Monte Gordo. Estas populações encontram-se isoladas e afectadas pela proximidade de povoamentos humanos e por repovoamentos com espécies invasoras, como são a acácia e os chorões, no caso da primeira, e o pinheiro, no caso da segunda.

Tabela 3. Estado das populações segundo os diferentes critérios de avaliação

Zona	**Localização Geomorfo-lógica**		**Densidade da população**		**Tamanho Indivíduo**		**Flora compa-nheira**		**Regene-ração Natural**		**Pressão humana**		**Estado das populações**	
Camariñas	Duna	0	Alta	0	Pequeno	0	Dunas	0	Baixa	2	Baixa	0	Bem	2
Caminha	Repovoamento	2	Baixa	2	Grande	2	Degradado	2	Média	1	Alta	2	Mal	11
São Jacinto	Duna	0	Alta	0	Pequeno	0	Dunas	0	Alta	0	Baixa	0	Bem	0
Mira	Duna	0	Alta	0	Médio	1	Dunas	0	Alta	0	Baixa	0	Bem	1
Peniche	Arriba	1	Média	1	Pequeno	0	Dunas	0	Média	1	Média	1	Com problemas	4
Caparica	Duna	0	Média	1	Médio	1	Dunas	0	Média	1	Média	1	Com problemas	4
Tróia	Duna	0	Alta	0	Médio	1	Dunas	0	Baixa	2	Baixa	0	Bem	3
Monte Clérigo	Arriba	1	Média	1	Médio	1	Dunas	0	Baixa	2	Média	1	Com problemas	6
Monte Gordo	Repovoamento	2	Baixa	2	Grande	2	Pinheiro	1	Baixa	2	Alta	2	Mal	11
Asperillo	Duna	0	Média	1	Médio	1	Dunas	0	Alta	0	Baixa	0	Bem	2
Bolonia	Repovoamento	2	Baixa	2	Médio	1	Pinheiro	1	Baixa	2	Baixa	0	Em regressão	8

Na figura 5, apresenta-se um resumo do estado das populações de *Corema album*, registando-se igualmente a fragmentação ou continuidade das populações. Os resultados mostram que só existem duas zonas com populações extensas de *C. album* apresentando indivíduos de todas as classes de idade, uma no litoral português entre Nazaré e Ovar, e a outra na costa de Huelva (Espanha) no sistema de dunas do Asperillo (Parque Natural de Doñana), existindo ainda outra população bem conservada nas dunas de Tróia. As outras populações estão limitadas a zonas de areia sobre arribas, como as populações da Costa Vicentina e Cabo Carvoeiro, ou estão reduzidas a fragmentos, destacando-se as populações da foz do Minho, de Monte Gordo, ou as de Bolonia na província de Cádiz. No caso da Galiza só existem duas populações desta espécie, uma no Parque Natural das Ilhas Cíes (não incluída neste estudo) e a outra em Camarinhas, que parece estar em expansão, e onde se observam restos de antigas populações formados por um ou dois indivíduos de grandes dimensões (FERNÁNDEZ DE LA CIGOÑA NUÑEZ, 1990).

Estas populações fragmentadas de *Corema album*, reduzidas a ilhas de areia sem regeneração natural, são formadas na sua maioria por indivíduos velhos, e a sua sobrevivência no futuro depende de medidas de protecção e regeneração de habitats.

Figura 5. Mapa das populações de *Corema album*, indicando o estado das mesmas e a localização de três populações não incluídas neste estudo.

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CHALLENGES FOR THE RECOVERY OF THE IBERIAN WOLF IN THE DOURO RIVER SOUTH REGION

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Habitat loss and fragmentation have been widely acknowledged as a primary cause of species decline, especially because isolated populations are more vulnerable to environmental and demographic stocasticity and local extinction. One good example of the effect of habitat fragmentation, at regional scale, is the present situation of the Iberian wolf in the Douro river south region. Historical persecution and, over the past few decades, increasing human population density and natural resource extraction activities have caused major changes to natural landscapes, and these have led to wolf population decline. Between 1995 and 1999 the wolf range was dramatically reduced and pack territories are currently disjunct. Protection of the present Iberian wolf range in the south of the Douro river region does not seem to be sufficient to reduce the local extinction risk. To evaluate wolf population viability for natural expansion, monitoring data collected over the last four years were used to investigate wolf habitat suitability, both inside and outside its present distribution area. The aim of this analysis is to assess and quantify the environmental factors affecting wolf habitat occupancy, as well as to formulate a predictive model to identify potential expansion zones. Results show that there is an area near the border with Spain which has good conditions for wolf occurrence and, in fact, is the most recent area where the Iberian wolf went extinct. However, this region is isolated from extant wolf populations. To decide on the best corridor to link this area with areas where wolves occur, the degree of connectivity between them is evaluated using the following aspects: present wolf distribution, human presence, habitat suitability and distance between suitable areas. Based on these results, areas of different conservation priority levels for the Iberian wolf in the south of the Douro river are identified, and conservation measures for each situation are suggested.

Keywords: Iberian wolf, south region of Douro river, habitat suitability, ecological corridors, conservation.

Grilo, C., Moço, G., Cândido, A. T., Alexandre, A. S. & Petrucci-Fonseca, F. (2002). Perspectivas para a recuperação do lobo-ibérico a sul do rio Douro. *Revista Biol. (Lisboa)* **20**: 121-133.

A redução e a fragmentação do habitat têm sido consideradas como as principais causas de declínio de muitas populações animais, que se tornam cada vez mais isoladas e vulneráveis aos efeitos negativos da estocasticidade ambiental e demográfica que potenciam o risco de extinção local. A actual situação do lobo-ibérico na região a sul do rio Douro é um bom exemplo deste fenómeno em virtude da fragmentação do seu habitat à escala regional. Nesta região, para além da histórica perseguição feita pelo Homem ao lobo, tem-se assistido nos últimos anos, ao aumento da densidade populacional humana, ao crescimento da rede viária e à sobrexploração de recursos naturais que têm provocado profundas alterações na paisagem. Entre 1995 e 1999 a área de distribuição do lobo a sul do rio Douro foi drasticamente reduzida, apresentando já evidentes linhas de fragmentação entre as alcateias. A protecção da actual área de ocorrência do lobo-ibérico não parece ser suficiente para garantir a preservação deste canídeo na região. Com base nos dados da monitorização do lobo, recolhidos nos últimos quatro anos, fez-se uma análise da adequabilidade do habitat para este carnívoro, dentro e fora da sua actual área de distribuição, visando avaliar as possibilidades de manutenção e expansão natural desta população. A análise consistiu na avaliação e quantificação dos factores ambientais que parecem afectar a distribuição do lobo, bem como no desenvolvimento de um modelo predictivo para identificar as áreas mais favoráveis para uma possível expansão. Os resultados obtidos neste estudo indicam que, para além da actual área de distribuição, a zona que apresenta mais condições para a sua ocorrência localiza-se na faixa junto à fronteira com Espanha, o mais recente local de desaparecimento desta espécie. No entanto, esta zona aparece isolada em relação à actual área de distribuição deste predador. Para seleccionar a melhor área de ligação entre estas duas regiões avaliou-se o grau de conectividade entre ambas considerando os seguintes aspectos: a actual presença de lobo, a presença humana, a adequabilidade do habitat e a distância entre as duas regiões.

Com base nos resultados anteriores foram ainda identificadas as áreas com diferentes níveis de prioridade de conservação do lobo-ibérico a sul do rio Douro, assim como as medidas a serem tomadas em cada caso.

Palavras-chave: lobo-ibérico, sul do rio Douro, adequabilidade do habitat, corredores ecológicos, conservação.

INTRODUCTION

Habitat loss and fragmentation have been widely acknowledged as a primary cause of species decline (PRIMACK, 1993). Habitat fragmentation generally leads to smaller and more isolated populations which are more vulnerable to local extinction due to stochastic events (GILPIN & SOULÉ, 1986). A good example of how habitat fragmentation at a regional scale can have an impact on wild populations is the present situation of the Iberian wolf (*Canis lupus signatus* Cabrera 1907) in the south region of the Douro river. In Europe, wolf population appears to have increased in the last years while in Portugal it continuous to decline. In Portugal, the Iberian wolf has been totally protected by law since 1988 (Law nº 90/88) and has the status "in danger of extinction" in the portuguese Red Book (SNPRCN, 1990). During the last century, the wolf population has decreased from South to North and from West to East. Besides the habitat reduction and fragmentation, the main causes of wolf population decline are human persecution and decreasing wild and domestic prey availability (PETRUCCI-FONSECA, 1990). In this study we present the results of the project "Definition of Ecological Corridors on the Conservation of Marginal and Fragmented Animal Population: the case of Southern Douro River Wolf Population" performed between 1997 and 1999. The progressive isolation and fragmentation of the wolf population in the southern Douro river region, with only seven packs currently presumed to be present, can only be prevented by creating ecological conditions for its maintenance and natural expansion. The aims of this research are: (1) analysis of habitat suitability, both inside and outside the present wolf distribution area in the south of Douro river region, (2) locate possible ecological corridors to link suitable areas, (3) identify priority areas for wolf recovery, (4) define conservation measures for each priority area and (5) evaluate the ability of the portuguese authorities to protect the wolf population in the southern Douro river region.

STUDY AREA

The study area is based on the wolf distribution in the 1970s in the southern Douro river region, according to PETRUCCI-FONSECA (1990) (Fig. 1). This study area includes zones where wolves went extinct recently that have not suffered significant habitat changes during the last 30 years and that may still have potential to be re-colonised by the species. The study area covers 20600 km^2 and encompasses the following administrative regions: Aveiro, Viseu, Guarda, Castelo Branco and Portalegre.

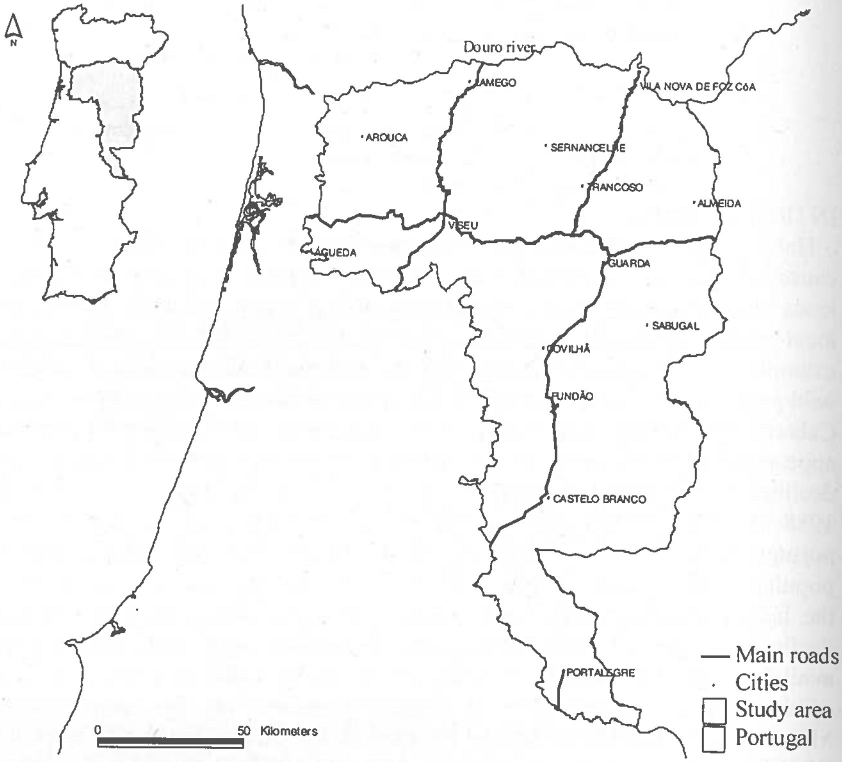

Figure 1 - Study area.

METHODS

Data set and scale

A geographic information system was used to compile the data on wolf occurrence and on the information described as relevant for wolf habitat (MASSOLO & MERIGGI, 1998, MLADENOFF *et al.*, 1999) (Tab. 1).

The software used was ArcView 3.1 (ESRI, 1996). The data on wolf occurrence were obtained by census and monitoring, from 1995-96 (ICN, 1997) through 1999. The sample unit considered to describe the study area is a 2x2km GAUSS square, which is a compromise between the pack size territories (DE LA VILLE *et al.*, 1997, MASSOLO & MERIGGI, 1998), environmental variables resolution and regional heterogeneity.

Habitat suitability analysis

The habitat suitability analysis includes the evaluation of factors that seem to affect wolf habitat occupancy and the development of a predictive model that can be used to identify areas presumed to be favourable for this predator.

Table 1. Data sources and origin of the environmental variables definition.

Source	Origin	Resolution	Year
Altitude	Direcção Geral do Ambiente	1/1 000 000	1982
Hydrology	Direcção Geral do Ambiente	1/1 000 000	1989
Land use	CNIG	1/25 000	1990
Land Cover Corine	CNIG	1/1 000 000	1985
Hunting zones	Direcção Geral das Florestas	1/1 000 000	1997
Human density	Instituto Nacional de Estatística	Regional administration	1991
Road density	Instituto Geográfico do Exército	1/250 000	1998
Livestock density	ADS	Regional administration	1999
Protected areas and Natura 2000 Network	ICN	1/1 000 000	1991
Administrative regions	Direcção Geral do Ambiente	Regional administration	1993

ADS – Agrupamentos de Defesa Sanitária; CNIG – Centro Nacional de Informação geográfica; ICN - Instituto de Conservação da Natureza.

The multiple logistic regression of wolf presence/absence data is the method selected for this analysis and has the following form: $\pi(x) = e^{g(x)}/(e^{g(x)}+1)$ where $\pi(x)$ is the occurrence probability of the species and g(x) is given by $g(x) = \beta_0 + \beta_1 x_1 + \beta_2 x_2 + ... + \beta_p x_p$, where β_0 is a constant and $\beta_1, \beta_2 .... \beta_p$, are the partial regression coefficients of the $x_1, x_2 ... x_p$ environmental variables (HOSMER & LEMESHOW, 1989). This procedure is considered very robust to violation of assumptions of parametric statistics such as normality and equal covariance-variance matrix (HOSMER & LEMESHOW, 1989).

In order to derive the predictive model the procedures suggested by HOSMER & LEMESHOW (1989) and BRITO *et al.*, (1999) were followed. Two sample squares were chosen: one to build the model and the other sample was used to validate the model. A proportion of 40% wolf presences (wolf presence between 1995-1999) and 60% absences (selected randomly in the study area) was used to avoid overfitting of coefficients and improve the predictive success of the model (BRITO *et al.*, 1999). The sample for validation included the remaining wolf presence sites that were deleted from the spatial autocorrelation minimisation (BRITO *et al.*, 1999) and wolf absence sites selected randomly.

Based on the model output, a wolf occurrence probability surface was built: high values of wolf presence probability correspond to suitable habitat for this carnivore.

Ecological corridor simulation

To identify the potential zone for wolf movement between the actual range and suitable areas for its expansion a simulation model with *CostDistance* extension tool for ArcView GIS 3.1 (ESRI, 1996) was used. The cost distance function requires a source grid (actual wolf range) and a cost grid (cost of passing through the pixel). The cost grid is the combination of the main roads, urban areas and the inverse of the habitat suitability model. That operation resulted in a surface with increasing cost values between 1 and 30: the lowest value represents suitable habitat for wolf and the higher values represent unsuitable habitat. Values between 1 and 10 were assigned to the inverse habitat suitability model and the value 10 to the main roads and urban areas. The result of this operation is a cost distance surface where each pixel is equal to the pixel distance multiplied by the average cost assigned until that pixel.

Identification of priority areas for the Iberian wolf recovery

Three levels of priority area are considered for the Iberian wolf south of the Douro river: *Priority I* - present wolf range; *Priority II* - ecological corridor area; *Priority III* - wolf suitable habitat where this species does not occur. To evaluate the real ability of the portuguese authorities to protect this carnivore in the suggested priority areas, the overlap between National Protected Areas (NPA) and Natura 2000 Network (NN) was taken into consideration.

RESULTS

Wolf habitat suitability model

The logistic model underlies the importance of three factors in determining wolf habitat suitability: forest cover, prey availability and human pressure (Tab. 2).

Table 2. Logistic regression model with the selected variables, coefficients (β) standard error (S.E.(β)) and p-value.

Environmental variables	β	S.E.(β)	p-value	ψ
Average altitude			0.000	
Altitude (300-600m)	2.326	1.072	0.030	10.241
Altitude (600-900m)	5.178	1.075	0.000	177.393
Altitude (>900m)	4.542	1.181	0.004	93.889
Mixed forest (presence/absence)	1.244	0.431	0.001	3.469
Livestock (nº/km^2)			0.000	
Livestock (43-97)	-1.583	0.438	0.000	0.205
Livestock (97-221)	-2.749	1.188	0.021	0.064
Livestock (221-526)	-5.768	15.212	0.705	0.003
Constant	-3.733	1.074	0.050	0.024

Altitude and mixed forest mainly account for wolf presence; otherwise high livestock densities have negative effects. The correct classification ratio for the presence is 78% and for absence is 90%. The model validation with a new sample is 86% of correct classification. An area is defined as "suitable wolf habitat" when occurrence probability is equal to or higher than 75% (Fig. 2). Figure 2 reveals that 23% of the study area provides suitable wolf habitat conditions, but the main suitable areas occur in two separate and isolated patches.

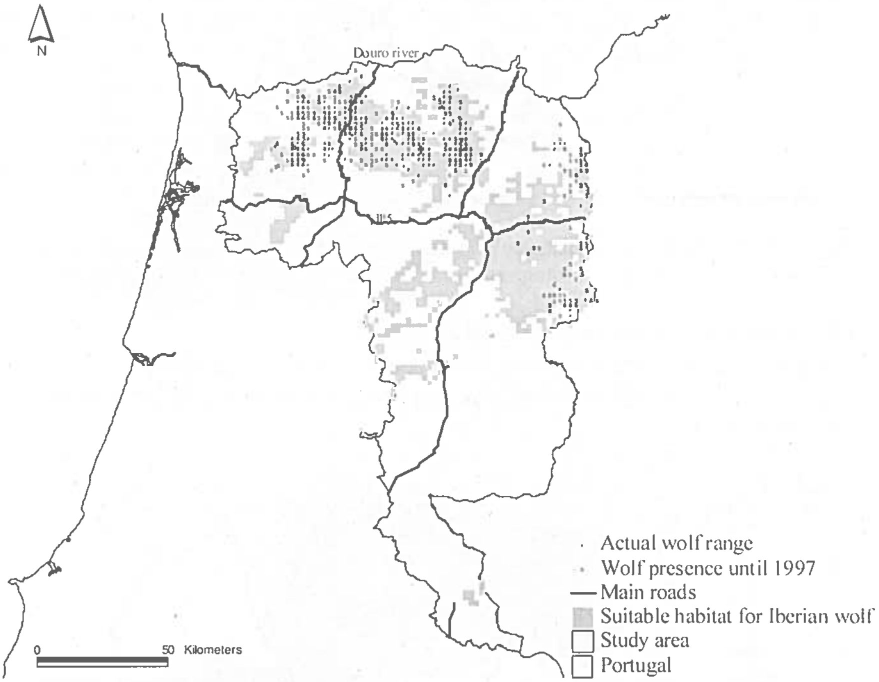

Figure 2. Distribution and suitable areas for the Iberian wolf in the southern Douro river region.

The western one corresponds to the current wolf range and shows evidence of significant developing fragmentation. The other patch is the region where the Iberian wolf became extinct most recently south of the Douro river.

Ecological corridor

The simulation results identified one potential site with minimum cost distance (40000-50000) linking the actual wolf range with suitable habitat for its expansion (Fig. 3). Analysing site habitat structure, it can be seen that a large part of that area includes permanent agriculture areas which wolves seem to avoid (MASSOLO & MERIGGI, 1998).

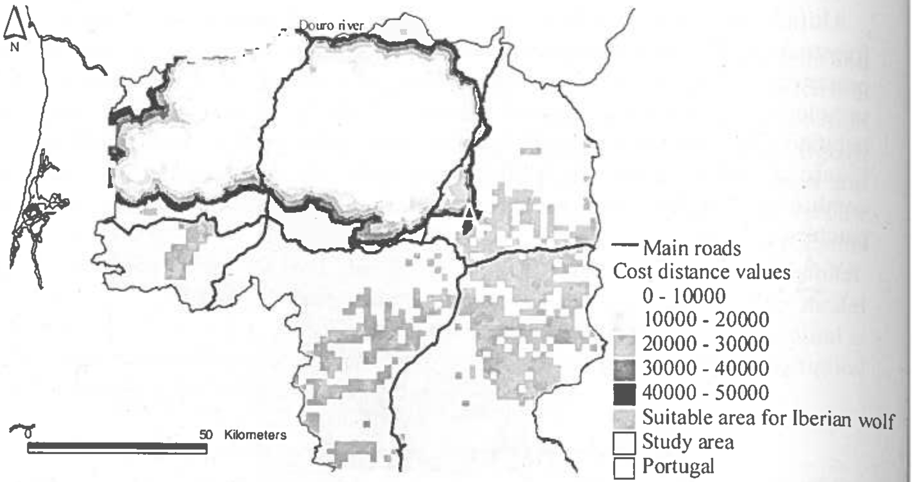

Figure 3 – Minimum cost distance simulation model for future ecological corridor (A) between present Iberian wolf range and suitable area for its expansion in the study area.

(3) Priority areas for Iberian wolf recovery

Figure 4 depicts areas with different conservation priority levels. Areas where the wolf occurs within the study area are suggested as first priority areas (14% of the study area).

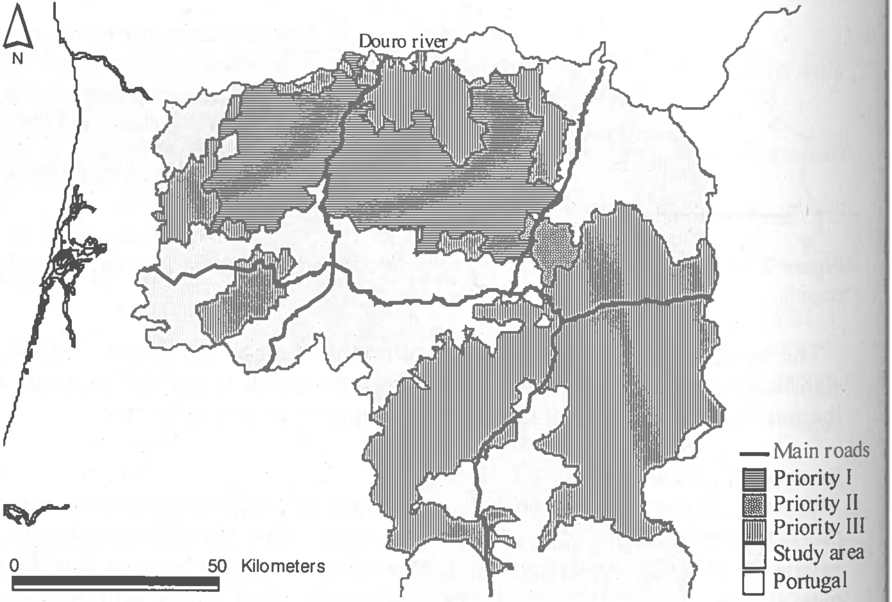

Figure 4. Priority areas suggested for the recovery of the Iberian wolf in the study area.

The second highest priority is assigned to the area where a future ecological corridor (140km^2) can be established and the third highest priority to potential areas for wolf expansion (30% of the study area). National protected areas and Natura 2000 Network cover about 37% of the study area (Fig. 5). Estrela Mountain Natural Park and the Malcata Natural Reserve are the only two protected areas included in the potential area for wolf expansion. In the western part of the present wolf range, Natura 2000 Network sites – Arada and Freita, Montemuro and Paiva river – are the only areas with legal protection. The Law does not protect the eastern part of the wolf range and the suggested ecological corridor.

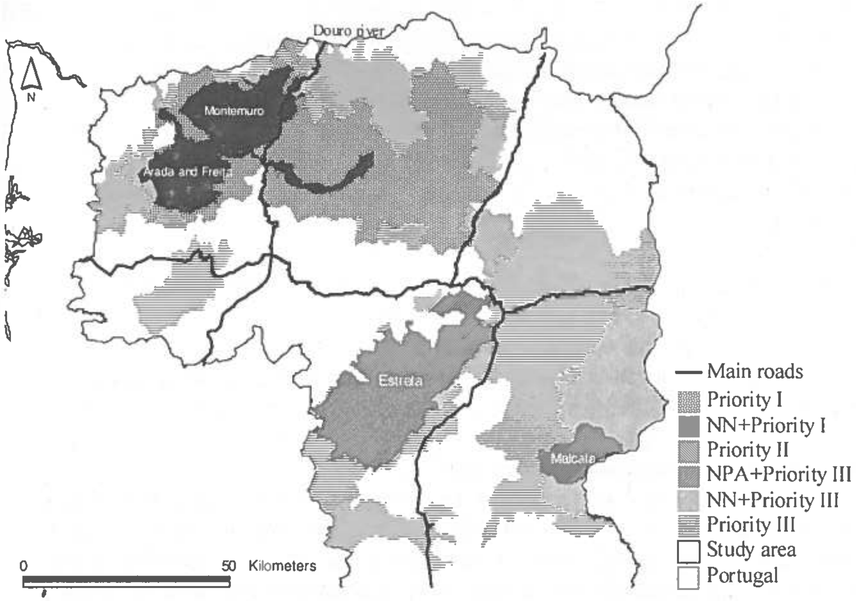

Figure 5. Overlap between suggested priority areas and both protected areas and sites included in the Natura 2000 Network.

DISCUSSION

The complexity of wolf social behaviour hinders any attempt to reduce wolf habitat choice and occupancy to a simple pattern of ecological attributes (MECH, 1970, MASSOLO & MERIGGI, 1998). Being top carnivores, wolves are not habitat–specific and are able to disperse over large regions, generally containing good and poor habitat for their establishment and successful breeding (MLADENOFF *et al.*, 1999). However, the future of this large carnivore, in a highly fragmented landscape mosaic such as occurs in Portugal, depends on our ability to establish extensive areas and connecting corridors (CORSI *et al.*, 1998) where it will be possible to ensure the viability of wolf populations.

According to the results, increasing isolation and fragmentation of suitable areas are threatening the wolf population survival in the south of the Douro river region. Fragmentation lines that could lead to local extinction in the short term are already affecting some packs. Forest cover constitutes an important factor for wolf occurrence; it provides suitable habitat for wild prey (in the study area only wild boar - *Sus scrofa* and a reduced population of roe deer - *Capreolus capreolus*) and reproduction and shelter (MECH, 1970). Anthropic pressure is the most likely explanation for the low occurrence/absence of wolves in low altitudes, where areas with high livestock densities suffer enhanced man-wolf competition for food and space (MASSOLO & MERIGGI, 1998).

The model developed shows two disjunct regions with similar habitat attributes which differ in terms of wolf presence. This situation can be explained by other factors that were not assessed in this study, such as the degree of wolf-man conflicts, public attitudes towards the wolf and wild prey distribution and abundance. These data were not available for this project and are probably important variables influencing the identification of suitable areas. The area identified as a possible ecological corridor needs direct intervention in terms of habitat management (restoration of forest cover) in order to become a real ecological corridor for the wolf.

ACTION LINES FOR THE RECOVERY OF THE IBERIAN WOLF IN THE SOUTH REGION OF THE DOURO RIVER

A summary of the relevant measures required for wolf recovery in the suggested priority areas is given in Table III. The measures are divided in three large action lines; some of the measures are already being adopted.

Habitat conservation and restoration

To conserve Iberian wolf packs in this region, the ecosystems they rely upon must be managed so that habitats are not lost nor their quality degraded. As mentioned previously, the habitat restoration in the identified possible ecological corridor is a fundamental step to link new expansion areas, allowing in the long term for wolf population viability. The classification of the suggested priority areas as protected areas (e.g. inclusion in the Natura 2000 Network) would be a strong tool in order to accomplish the goals of the action lines. Even if is not possible to legally protect the suggested priority areas, it would be extremely important to elaborate a regional conservation strategy and ask for co-operation from all authorities interested in or affected by the presence of the wolf. Public and private entities such as Instituto da Conservação da Natureza (Nature Conservation Institute), Instituto Florestal (Forest Institute), Instituto das Estradas (Roads Institute), hunting organisations, livestock owners and regional authorities should seriously be involved in the elaboration of the conservation strategy.

Table III. Suggested conservation measures for priority areas (PA) and the entities already implementing them.

Conservation measures	PA	Implementation	Entities
Conservation/restoration habitat	I II III	Yes (30%)	ICN
Public attitudes toward wolf studies	I II III	Yes	*
Educational program	I II III	No	-
Campaign against poisoning and snaring	I II III	No	-
Wild prey re-introduction program	III	Yes	DGF**
Livestock guarding dogs	I III	Yes (I)	ICN &***
Compensation for wolf depredation	I II III	Yes	ICN
Wolf population research and monitoring	I	Yes	*

* "A População Lupina a sul do rio Douro em Portugal: análise temporal, atitudes públicas e aperfeiçoamento dos corredores ecológicos" funded by Fundação da Ciência e Tecnologia in program SAPIENS 99 e coordenated by Grupo Lobo and Centro de Biologia Ambiental.

** Direcção Geral de Florestas.

*** "Adaptação de Sistemas Tradicionais de Protecção dos Rebanhos" para a Conservação da Natureza funded by Fundação da Ciência e Tecnologia, Instituto da Conservação da Natureza and coordenated by Grupo Lobo and Centro de Biologia Ambiental.

Man-wolf conflict reduction

The man-wolf relationship is mainly based on livestock damage and beliefs that make the wolf an imaginary evil and dangerous animal for Man. To minimise public animosity and wolf persecution the following actions are needed:

(1) man-wolf interaction studies about the reasons, people profile and geographic localisation of the conflicts as a basis to outline educational programs;

(2) continuous and widespread public educational efforts and campaigns against poisoning, snaring and other ways of killing wolves have proven to be relevant in changing attitudes toward wolves resulting in decreased killing. Authorities must be persuaded to implement the current wolf protection law;

(3) decrease wolf predation impact on livestock by reintroducing wild prey and promoting the use of livestock guarding dogs;

(4) political pressure to assure earlier compensation payments for wolf damage to livestock;

Wolf research and monitoring

A research programme should be co-implemented along with other guidelines suggested because it can contribute important information to the planning of conservation strategies. Research should include several aspects of wolf ecology namely pack size, diet, hunting behaviour, habitat use, wolf movements and activity. These studies can provide information on wolf distribution, wolf use of the area suggested for an ecological corridor, the impact of human settlements on wolf movements and on the factors affecting wolf dispersal.

CONCLUSIONS

Modelling priority areas can facilitate fieldwork research efforts and may provide useful tools to improve objectivity and efficiency of conservation strategies (GLENZ *et* al, 2001). The suggestions presented here should not be taken as immutable nor final. These results are part of a data improvement and actualisation process that should be continuous and cyclic. With new data and information it will be possible to identify new areas besides the ones suggested here. The suggested guidelines for priority areas management require involvement from local populations and a reasonable financial investment. Due to the complexity of the conservation of the Iberian wolf in Portugal, its recovery in the south region of the Douro river is a big challenge for the governmental agencies that have undertaken the responsibility for the conservation of this emblematic species.

ACKNOWLEDGMENTS

This study was supported by PRAXIS XXI Programme (PRAXIS XXI/PCNA/BIA/00108/96). We are thankful to Eduardo Cruz for technical support and CNIG for logistic support. We also would like to thank Francisco Álvares, José Carlos Brito for their comments and Isabel do R. Amorim and Dr. Johnny Birks for suggestions on the english version.

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USE OF MIDDENS BY THE COMMON GENET (*GENETTA GENETTA* L.) AND ITS RELATION WITH THE LANDSCAPE STRUCTURE IN GRÂNDOLA MOUNTAIN (SW PORTUGAL)

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Costa, H. & Santos-Reis, M. (2002). Use of middens by the common genet (*Genetta genetta* L.) and its relation with the landscape structure in Serra de Grândola (SW Portugal). *Revista Biol. (Lisboa)* **20**: 135-145.

Forty-three middens of common genet, *Genetta genetta*, were visited on a daily basis in order to evaluate their use rate. The *k*-means method was used to classify the middens into two groups in terms of their intensity of use (moderate and high). A land use map of the study area was prepared and the area surrounding twenty-two previously selected middens was analysed using landscape variables. A logistic regression was conducted to build up a model that could predict which variables had an influence on the use of the middens by the common genet. Only one variable – Mean patch size of cork oak woodland without understorey (MPCWU) was selected to the model. The model fit was significant, with a high number of correct classifications (90. 9%). Middens surrounded by larger cork oak woodland patches without understorey are less utilised. This behaviour is maybe due to the scarcity of food and shelter provided by this habitat.

Keywords: *Genetta genetta*, marking behaviour, Mediterranean landscape, logistic regression.

Costa, H. & Santos-Reis, M. (2002). Relação entre o grau de utilização de latrinas por geneta (*Genetta genetta* L.) e a estrutura da paisagem na Serra de Grândola (SW Portugal). ***Revista Biol. (Lisboa)*** **20**: 135-145.

Foi avaliado o grau de utilização de 43 latrinas de geneta, *Genetta genetta*, através da sua monitorização diária. Utilizou-se o método das *k*-médias para agrupar as latrinas em 2 grupos consoante o seu grau de utilização (moderado e elevado). Elaborou-se uma carta de uso do solo da área de estudo e caracterizou-se, com base em variáveis de estrutura da paisagem, a área envolvente a 22 latrinas previamente seleccionadas. Recorreu-se a uma análise de regressão logística para elaborar um modelo que permitisse prever quais os factores, a nível de paisagem, que influenciam o grau de utilização de latrinas por parte da geneta. A variável área média de manchas de montado sem sub-coberto (MPCWU) foi a única seleccionada para o modelo. O modelo ajusta-se adequadamente aos dados, sendo a proporção de casos correctamente classificados de 90. 9%. Conclui-se que as latrinas localizadas em maiores áreas de montado sem sub-coberto são as menos utilizadas, podendo este facto explicar-se pela reduzida importância deste biótopo em termos de refúgio e de recursos tróficos.

Palavras chave: *Genetta genetta*, comportamento de marcação, paisagem mediterrânica, regressão logística.

INTRODUCTION

The dominant mode of mammalian communication is olfactory (THIESSEN & RICE, 1976). In this respect, it plays an important role in their social behaviour, particularly in territorial mammals where scent-marking may constitute the key element of territory maintenance (BARRETE & MESSIER, 1980). It is widely accepted that many mammals scent mark their territories in order to advertise their occupation and ownership of the territory (ERLINGE *et al.*, 1982; RICHARDSON, 1991), however, this is just one of its functions. It is also a means of exchanging information, orienting the movement of individuals, and integrating social and reproductive behaviour (EISENBERG & KEIMAN, 1972).

In several carnivore species, faeces are usually deposited at particular locations known as middens (ROPER *et al.*, 1993), which serve an important function as scent-marking sites (RICHARDSON, 1990). The common genet, *Genetta genetta* L. (Carnivora: Viverridae), is one of the species that exhibits this kind of behaviour (ROEDER, 1980; PALOMARES, 1993).

In several studies involving carnivores, variations in marking behaviour were attributed to habitat changes (MACDONALD, 1980). In a study with common genets at Doñana National Park (Spain), it became evident that the dissimilarities regarding marking behaviour by this species were explained by the differences observed in habitat composition (PALOMARES, 1993).

The main aim of this study is to analyse the changes in the marking behaviour of the common genet in an area strongly exposed to an anthropogenic influence like that found in the cork oak woodland (further on named as cork “montado”) of Serra de Grândola (CORREIA & NISA, 1999). By complementing the researches made by ESPÍRITO SANTO (1999) and COSTA (2000), we aim to analyse the relation between the intensity of use of the common genet's middens, their location, and the structure of the landscape.

STUDY AREA

The study area of 22,3 km^2 is located on the eastern slopes of Serra de Grândola (SW Portugal), and includes a 221ha property (Herdade da Ribeira Abaixo) where the field station of the Environmental Biology Centre (CBA) is located.

Altitude ranges from 150 to 270 m. The area is crossed by several temporary streams and by a perennial one (Rib^a dos Castelhanos). The climate is of mediterranean type, with some atlantic influence, characterized by hot, dry summers and mild winters.

The dominant tree species is the cork oak (*Quercus suber*) but the holm oak (*Quercus ilex*) and the Lusitanian oak (*Quercus faginea*) are also present. The scrubland is mainly composed of *Cistus* spp., French lavender (*Lavandula stoechas*), strawberry tree (*Arbutus unedo*) and greenweed (*Genista triacanthus*) (CORREIA & NISA, 1999). Extremely dense patches of spontaneous vegetation (Mediterranean maquis) can be found on some slopes where human intervention is low. Riparian corridors prevail near watercourses and are dominated by alders (*Alnus glutinosa*), poplars (*Populus nigra*), willows (*Salix atrocinerea*) and hawthorns (*Crataegus monogyna*).

Human density is low and the activity that causes most impact on the floristic and faunistic communities of the region is undoubtedly the practice of cork removal, which implies brushcutting, opening roads and the lopping of cork oak branches (SANTOS-REIS & CORREIA, 1999).

MATERIAL AND METHODS

Survey and monitoring of the middens

An exhaustive search of middens was made in the study area during September 1999, and their monitoring was carried out between October 1999 and June 2000. During this period the middens were visited for 13 consecutive days in the beginning of each month. Every sampling day, including the first, all the faeces found in each midden were identified, counted and crushed. Faeces found on each first's day visit were not considered in the calculations of the daily visits by genets to middens. Special concern was taken to not to leave any olfactory clues.

Landscape analysis

Aerial photographs taken in 1995 (1:10 000) and the ArcView 3.1 GIS software (ESRI, 1996) were used to build up a digital map of the landscape that was subsequently ratified by an intensive field survey in order to detect recent land use changes.

The resulting map (Fig. 1) distinguishes 11 landscape units or habitats: eucalyptus groves with and without understorey; orchards; olive groves;

Mediterranean maquis; riparian vegetation; grasslands; cork "montado" with very dense understorey (less than 30% of bare ground); cork "montado" with dense understorey (between 30 and 60% of bare ground); cork "montado" with little understorey (more than 60% of bare ground) and cork "montado" without understorey.

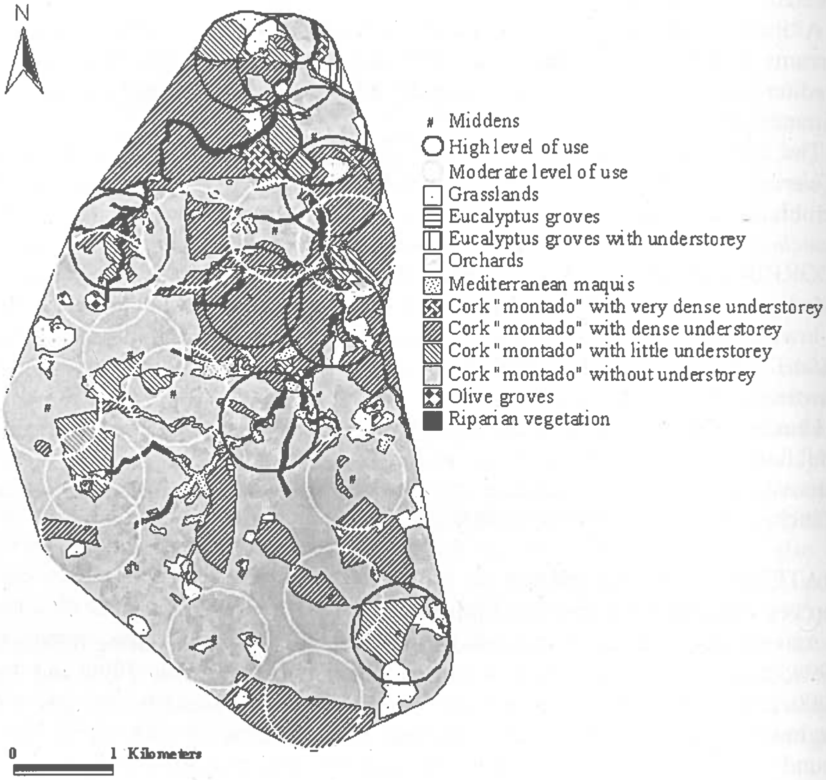

Figure 1. Land use map of the study area and location of the buffer zones surrounding each of the 22 middens selected to the statistical analysis.

Selection and description of the middens

The precise location of the middens (data obtained through a Global Positioning System) was overlaid onto the land use map, and a buffer zone with a 500 m radius was created in order to characterize the structure of the surrounding landscape. When two middens were close to each other and their buffer zones overlapped more than 50%, the less used was disregarded. This option intended to avoid the pseudoreplication effect (HURLBERT, 1984).

The Patch Analyst extension (ELKIE *et al.*, 1999) of ArcView 3. 1 software was used to analyse the structure of the landscape in each buffer zone. Several landscape parameters were quantified (number of habitats, number of patches, mean patch size, Shannon's diversity and evenness indexes, and, for each habitat, the total area, the number of patches and the mean patch size), which resulted in 47 variables.

Middens intensity of use

The middens level of use was quantified by the *Visit Rate* (VR – number of visits in which faeces were found / number of visits to the midden), the *Daily Use Rate* (DUR - total number of faeces found in the midden / number of days in which faeces were found in the midden) and the *Total Number of Faeces* found in each midden (TNF).

Statistical analysis

A cluster analysis was conducted with the three use-related variables, in order to group the middens according to their intensity of use. The *k*-means method was used, but first the variables were standardized due to the wide dispersion of their values. This procedure allowed for the equal contribution of the three variables to the calculation of the similarities between the middens (ZAR, 1996).

Logistic regression has been employed to predict species distributions (BRITO *et al.*, 1999), habitat preferences (PALMA *et al.*, 1999) and abundance/densities (SCHONROGGE *et al.*, 1996). In this study, the logistic regression was used to evaluate the influence of landscape features on the use of the middens by the common genet.

In a previous step it was necessary to select the variables that would be considered in the model's development. Wald and Mann-Whitney tests were used to examine the relation between the independent and dependent variables, the former only being selected when the *p*-value associated with both tests was less than 0. 25 (HOSMER & LEMESHOW, 1999).

The logistic regression is extremely sensitive to multicollinearity (TABACHNICK & FIDELL, 1996), arising when independent variables are inextricably associated, making it impossible to accurately distinguish their individual contributions to the model (NORTH & REYNOLDS, 1996). In order to overcome this problem the Spearman's rank coefficient (r_s) (ZAR, 1996) was calculated for all the possible pairs of variables selected for the model. When a pair of variables was considered highly correlated ($r_s \geq |0.\ 7|$), the most biologically plausible or the one that better expressed the characteristics of the study area was selected.

The variables retained in the previous selection were subsequently applied to a forward stepwise procedure. The model was used to predict the probability of a successful classification of the middens considering its intensity of use (dependent variable). Independent variables entered into the model at the 0. 15 significance level and were removed at the 0. 20 level, using the likelihood-ratio test (PEREIRA, 1996). Wald test was used to detect which variables significantly contributed to the model and only those with a score of p < 0. 05 were retained (HOSMER & LEMESHOW, 1989).

The goodness-of-fit accuracy was assessed by the Pearson χ^2 and by the correct classification rates (HOSMER & LEMESHOW, 1989).

The final step of the logistic regression consists in the validation of the model. In other words, it is necessary to evaluate its predictive power using another sample (HOSMER & LEMESHOW, 1989). Because the initial middens sample was small it was considered convenient to use the jackknife technique (KREBS, 1989). In this procedure one midden is removed from the initial data and the remaining are used to estimate the model parameters. The fitted model is then applied to this unused midden to predict its degree of use. This process is repeated for each midden in the dataset, giving an unbiased estimate of the classification error for each use class (NORTH & REYNOLDS, 1996). This technique also allows the evaluation of the model fit and therefore its extrapolation capacity, resulting in the analysis of the variables selected for the different models obtained.

RESULTS

Intensity of use

The method used to minimize pseudoreplication resulted in the selection of 22 of the 43 middens detected and subsequently sampled (COSTA, 2000). Figure 2 shows the standardized values of the variables TNF, VR e DUR for each midden. Because the first two were highly correlated (r_s = 0. 99; n = 22; p < 0. 001) only one of them was included in the *k*-means procedure (VR).

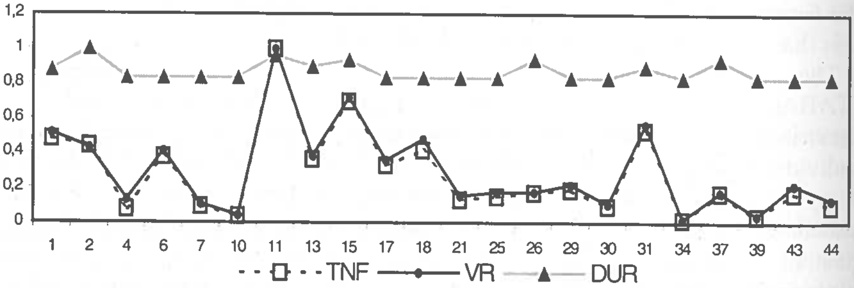

Figure 2. Standardized variables - Total number of faeces (TNF), Visit Rate (VR) and Daily Use Rate (DUR) by midden.

The difference in the middens intensity of use allowed the classification in two groups (Tab. 1): one corresponding to a moderate intensity of use (group 0; n=13) and the second to a high intensity of use (group 1; n=9).

Table 1. Cluster (group) means, values and significance of the F test.

Variable	Group 0	Group 1	F	p
VR	0. 130	0. 539	46. 231	$1x10^{-6}$
DUR	0. 849	0. 896	4. 994	0. 037

Statistical analysis

From the 47 original variables, 20 were significant by the Wald and Mann-Whitney tests. However, after testing for multicollinearity, only 6 were retained for further analysis (Tab. 2).

Table 2. Significance values of the Wald (W) and Mann-Whitney (M-W) tests for the variables selected for the multivariate analysis (A. grass. – Total area of grassland; A. rip. veg. – Total area of riparian vegetation; SEI – Shannon's evenness index; MPCDU – Mean patch size of cork oak woodland with dense understorey; MPorchs. – Mean patch size of orchards and olive groves; MPCWU – Mean patch size of cork oak woodland without understorey).

	A. grass.	A. rip. veg.	SEI	MPCDU	MPorchs.	MPCWU
W	0. 147	0. 041	0. 203	0. 241	0. 109	0. 048
M-W	0. 119	0. 040	0. 192	0. 214	0. 094	0. 000

The forward stepwise selection created the more parsimonious model having only one variable (MPCWU – Mean patch size of cork "montado" without understorey – Tab. 3). From a cut-off point of 0. 5 up, the model classified the middens into group 1. The results of this analysis (90. 9% correct classifications) and the statistical tests conducted proved that the model has a good fit, registering a fine balance between the correct and incorrect classification rates (Tab. 4).

Table 3. Model obtained by the logistic regression procedure (β - estimated coefficients; S. E. (β) – standard errors of the estimated coefficients; Wald – results of the Wald test; Wald (P) – significance of the Wald test).

Variable	Coefficient β	S. E. (β)	Wald	Wald (p)
MPCWU	-0. 202	0. 102	3. 904	0. 048
Constant	3. 177	1. 600	3. 943	0. 047

Table 4. Pearson χ^2, Hosmer-Lemeshow Tests, and correct classification rates results with a cut-off point of 0. 5.

Pearson χ^2			**Correct classification rates**		
χ^2	g. l.	p	Group 0	Group 1	Total
16. 705	22	0. 779	92. 3 %	88. 9 %	90. 9 %

All the results obtained with the jackknife technique were consistent. The MPCWU was the only variable selected in each model. Results of the evaluation of the model quality using this technique showed that the percentage of correct classifications was 91. 7 and 80. 0 for group 0 and 1 respectively. Only 8. 3 % of the middens were incorrectly classified into group 0 while they should have been into group 1, and 20% into group 1 while they should be grouped into 0 (Tab. 5). In a global analysis, only 13. 6% of the middens were incorrectly classified.

Table 5. Correct classifications and errors (italic) obtained by the jackknife technique.

Level of use	Expected 0	Expected 1
Observed 0	91. 7 %	8. 3 %
Observed 1	20. 0 %	80. 0 %

Although the variable *area of riparian vegetation* (A. rip. veg.) was not selected for the final model, the results of the univariate analysis suggest its probable and potential importance (Tab. 1). A new logistic regression model was then developed excluding the variable MPCWU. The variable A. rip. veg. was the only one selected with a positive β coefficient and a Wald Test significance of 0. 041. In this case, as suspected, the correct classification rate was lower (77. 3%) than the one obtained with the first model.

DISCUSSION

The selection of the variable mean patch size of cork "montado" without understorey with a negative β coefficient, and also the good fit and high classification rates allowed by the model, shows that the common genet makes less use of the middens when surrounded by large patches of this habitat. MACDONALD (1980) refers that in areas where the habitat interpenetration is high, it is expected that the distribution of the middens coincide with the one of the defended resources. The logistic regression analysis that omitted the variable MPCWU, showed that the most actively used middens were surrounded by larger areas of riparian vegetation. Although this result must be considered with caution (because the variable MPCWU is being ignored), it is consistent with the results obtained by LOURENÇO (1999). She verified that this species strongly prefers areas of riparian vegetation, which are minimally expressed in the study area (2. 5% of the land use).

These results can be explained by looking at the characteristics of the aforementioned habitats. Not only do the patches of cork "montado" without understorey provide little shelter (limited to the higher trees), but they also provide much less food.

In the study area, the two main food sources for the common genet (as regards consumed biomass) are small mammals (nearly 30%) and fruit (25%) (SANTOS & PINTO, 1998). These authors confirmed that the abundance of small mammals was reduced in the patches of cork "montado" without understorey with 62 animals captured in 2000 trap-nights. This value contrasts with the one obtained for patches of cork "montado" with understorey where 119 animals were captured in 2000 trap-nights. In the patches of riparian vegetation the difference was even greater with the capture of 190 animals in 2000 trap-nights. As regarding fruit availability they are almost absent from the cork "montado" without understorey while, on the other hand, fruit is abundant in the areas of riparian vegetation (blackberries, strawberry tree fruit, wild pears and figs).

PEREIRA (1999), verified that, in the study area, the patches of dense shrubby vegetation (specially near the watersheds), not only included 30. 8% of the 91 rest places used by 3 radio-tagged common genets but also corresponded to the places where these animals were mostly detected while active. The higher importance of this habitat is confirmed by results obtained by this author, who detected a strong association between its use and the period in which the animals were rearing their young.

The different use of middens can also be explained by the frequent brushcutting activities conducted in the study area. When areas rich in food and shelter are cut, they loose their importance to the common genet, often resulting in a decrease in the use of the middens (COSTA, 2000). Observations made by ROEDER (1980) may explain why they are not completely abandoned: he verified that midden's use by the common genet is also dependent on a habituation process.

The selection of only one variable to the model (MPCWU) may also be influenced by the high representativeness (54. 2% of the land use) and mean patch size (0. 8 Km^2) of this habitat in the study area. The lower representativeness of the other habitats and their high interpenetration may dissemble their real importance for midden's use by the common genet.

To better explore these circumstances we propose, in a future work, the use of smaller buffers around each midden, which should induce an improved perception of the factors that regulate its use. However, it should also be stressed that the reduced number of middens substantially influences the logistic regression results.

In order to understand the real importance of habitats like the riparian corridors and the Mediterranean maquis for the common genet, more studies are needed.

These will certainly provide extremely important data for the correct management of cork "montado" areas. This might imply, for example, leaving some areas uncut during brushcutting operations at the time of cork harvesting.

ACKNOWLEDGEMENTS

We are grateful for the logistic support supplied by the Environmental Biology Center (CBA), which proved essential to the accomplishment of this study. We thank Clara Grilo and José Carlos Brito for sharing experiences in the statistical analysis. We are especially grateful to Maria João Cruz and Edwin Leander for all the help that they so enthusiastically gave in the field and also to Simon Wates for the grammatical revision.

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FIRST DATA ON THE SOCIAL AND SPATIAL STRUCTURE OF AN EURASIAN BADGER (*MELES MELES* L., 1758) POPULATION IN A CORK OAK WOODLAND (SW PORTUGAL)

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Rosalino, L. M., Loureiro, L, Santos-Reis, M. & Macdonald, D. W. (2002). First data on the social and spatial structure of an Eurasian badger (*Meles meles* L., 1758) population in a cork oak woodland (SW Portugal). *Revista Biol. (Lisboa)* **20**: 147-154.

The Eurasian badger (*Meles meles*) is a social carnivore with a widespread European and Asian distribution. Nevertheless, the majority of the studies regarding this mustelid were developed in NW Europe, especially in the UK. In the last decade other projects were conducted outside UK, in order to understand the life history adaptations of this social predator to different environments (*e. g.* Swiss alps and Mediterranean areas such as Spain and Italy). The present study aims to develop an understanding of which adaptations the badger has developed in order to survive in a Mediterranean habitat – i. e. the cork oak woodland – a multiple use agro system where agriculture and cattle raising practices are associated with high biodiversity. After intensive prospecting, all latrines and badger setts found were further on monitored in order to evaluate the degree and pattern of use. Badgers were captured, marked and some radio-tracked. The simultaneous analysis of these data led us to the identification of four social groups in an area of 35 km^2, each composed by six to eight individuals (three/four adult and three/four cubs/juveniles). Although home ranges vary between individuals and members of a social group, they show a high degree of intra-group overlap and exclusiveness in relation to neighbouring groups. Nevertheless, during the mating season at least one adult male made an incursion into the territory of a neighbouring group. In SW Portugal, an area dominated by cork oak woodlands, badgers although living in low density seem to be organized in well-defined social and spatial units.

Keywords: Badger, *Meles meles,* Spatial ecology, Home range, Social structure, Mediterranean region.

Rosalino, L. M., Loureiro, F., Santos-Reis, M. & D. W. Macdonald. (2002). Primeiros dados sobre a organização social e espacial de uma população de texugos (*Meles meles* L., 1758) num montado de sobro (SW Portugal). *Revista Biol. (Lisboa)* **20**: 147-154.

O texugo (*Meles meles* L., 1758) é um carnívoro social com ampla distribuição europeia e asiática. No entanto, a grande maioria dos estudos efectuados sobre este mustelídeo foram realizados no NW da Europa, principalmente em Inglaterra. Nos últimos anos têm sido desenvolvidos diversos trabalhos noutras áreas da sua distribuição, com o objectivo de conhecer as estratégias de vida deste predador social em ambientes tão diferentes como os Alpes Suíços e as zonas mediterrânicas (*e. g.* Espanha e Itália). O presente estudo visa conhecer as adaptações desenvolvidas pelo texugo para conseguir sobreviver num habitat mediterrânico com características muito particulares - o montado de sobro - um agrossistema de uso múltiplo, onde a utilização agro-silvo-pastoril está aliada a uma elevada biodiversidade. Após uma prospecção intensiva da área de estudo, para a detecção de tocas e latrinas, procedeu-se à monitorização das mesmas, para determinar o respectivo grau e padrão de utilização. Seguidamente capturaram-se animais adultos para posterior radio-rastreio. A análise conjunta destes dados possibilitou a identificação de quatro grupos sociais distintos, numa área com cerca de 35 km^2, constituídos em média por seis a oito indivíduos (três a quatro adultos e três a quatro crias/juvenis). Apesar da dimensão das áreas vitais apresentar variações inter-indivíduos, estas tendem a ser sobrepostas intra-grupos e exclusivas em relação a grupos sociais vizinhos. No entanto, na altura da reprodução verificou-se que, pelo menos uma vez, um macho adulto efectuou uma incursão no território de um grupo social vizinho. Nesta região do SW de Portugal, dominada por vastas áreas com montado de sobro, a população de texugos residentes, apesar de apresentar baixa densidade, aparenta ter uma estrutura social e espacial bem definida.

Palavras chave: Texugo, *Meles meles*, Ecologia espacial, Áreas vitais, Estrutura social, Região mediterrânea.

INTRODUCTION

The Eurasian badger (*Meles meles* L., 1758) is a mesocarnivore with a wide distribution that includes areas as different as English pastures, Finish boreal forests, Mediterranean woodlands and Chinese steppes (MITCHELL-JONES *et al.*, 1998). In spite of this broad distribution, the scientific knowledge regarding its ecology results from data collected in NW Europe, mainly England, where the species live in high density (*e. g.* KRUUK, 1978; MELLGREN & ROPER, 1986; SHEPHERDSON, *et al*, 1990; TUYTTENS *et al.*, 2000) is still scarce. The singular biophysical characteristics of the Mediterranean regions, particularly in what concerns the high degree of humanisation of the landscapes, justify the need to evaluate the ecological adaptations of this carnivore to those special conditions. Thus, a project was developed aiming to characterize this predator's life strategy in Portuguese cork oak woodlands.

In this paper we intend to illustrate the social and spatial organization of the badger population that inhabits the study area, namely:

- to confirm if the individuals show a social pattern similar to those described for north and north-western European populations, with stable and well defined social groups (NEAL & CHEESEMAN, 1996) or if they have a solitary behaviour as was recorded in other Mediterranean regions (PIGOZZI, 1987):
- to evaluate home-ranges and core-areas sizes, and their spatial distribution.

STUDY AREA

The study was developed in Baixo Alentejo (Southwest Portugal), in a 35 km^2 area located on the eastern slope of Grândola Mountain. This area is included in the Sado river basin and is crossed by a permanent stream (Ribeira de Castelhanos) and several with intermittent flow. Altitude varies between 159m and 238m, and the climate is Mediterranean with Atlantic influence, with a mean annual temperature of 15°C to 17,5 °C (CORREIA & SANTOS-REIS, 1999).

The region is characterized by a forest dominated by cork oaks (*Quercus suber*), associated some times with holm oaks (*Q. ilex*) and Lusitanian oaks (*Q. faginea*). The dense understory is composed by mixed shrubs (*Cistus* spp., *Lavandula stoechas*, *Arbutus unedo*, *Erica* spp.). Near the streams the riparian vegetation included common alders (*Alnus glutinosa*), black poplars (*Populus nigra*), common sallows (*Salix atrocinerea*) and blackberry bushes (*Rubus ulmifolius*). Tasmanian blue gum (*Eucalyptus globulus*) and maritime pine trees (*Pinus pinaster*) can also be found in the study area, although only in small patches. Where the understory was cut, pastures with gramineous plants such as *Aira caryophyllea* and *Briza* sp. can be found (CORREIA & SANTOS-REIS, 1999).

The study area includes cork oak woodlands, that have been abandoned since 1965, being human activities reduced only to cork extraction, cattle raising (with small herds), hunting and traditional agriculture (CORREIA & SANTOS-REIS, 1999).

METHODS

This project started in 1999, and is still ongoing. As a first step the study area was surveyed with the objective of mapping the species signs of presence (latrines, footprints and setts). The identified latrines and setts were monitored fortnightly and monthly, respectively, to evaluate their use rate. In order to characterize the structure and demography of the population (sexual and age ratios, reproduction, etc), and to radio-tag some animals, a capture program was implemented. The selection of the capture sites was based on the use rate of setts.

Of all the captured animals, those that reached adult size were selected and fitted with a radio-transmitter (BIOTRACK, UK). After release they were regularly located and their home-ranges and core-areas determined, using the Minimum Convex Polygon Method with 100% and 50% of the total locations, respectively (KENWARD, 1987; WHITE, & GARROTT, 1990).

RESULTS AND DISCUSSION

The intensive surveillance of the study area allowed the detection of 29 setts and 87 latrines. In all located setts, at least once, latrines with fresh scats were encountered. The two years sett monitoring allowed the identification of those that were more frequently used. Thus, only 11 showed use rates higher than 50%, which indicates that they could be important diurnal resting sites, and so were selected to implement the capture program. The 87 latrines showed highly variable use rates that didn't allow, through an exploratory analysis, the identification of a spatial distribution pattern.

During the capture program (from February 2000 to August 2001) 27 captures of 22 different animals were achieved: 10 adults and 12 juveniles (Table 1), in a total of 1047 night/traps, meaning that it was necessary to have a trap active for 38. 8 nights in order to capture one badger. The adult sex-ratio was 1M:1. 5F, similar to the ones described for the northern populations (NEAL & CHEESEMAN, 1996). Nevertheless, when adults and juveniles are considered together the sex-ratio equals males and females. This fact can be interpreted as differential juvenile mortality between sexes, or with a higher tendency of males to disperse (CHEESEMAN *et al.*, 1988; NEAL & CHEESEMAN, 1996).

From the 13 setts involved in the trapping programme (two selected for their geographical location and 11 for their higher use rates) only in five bagers were capture (Tab. 1). It was possible to confirm reproduction in two of those setts (Aroeira and Fetos) in the year 2000, with four pups captured in each one, and in 2001 (Aroeira and Amadas), with three and one pup captured, respectively. The number of pups *per* year and *per* social group was similar to those obtained by other authors in different regions of Europe (*e. g.* KRUUK, 1989).

The analysis of figure 1 shows the existence of three well defined social groups: i) the northern group, with three radio-tagged animals whose homeranges are almost perfectly coincident; ii) the southeast group, with three

Table 1. Captured animals in 1999, 2000 and 2001

Sett	Social group	Year	Adults		Juveniles		Mark*
			♂	♀	♂	♀	
Adrejão	SW	2001		1			1♀
Caniçais	SW	2000	1				
		2001	1				1♂
Amadas	SE	2001				1	
Aroeira	SE	2000	1	1	3	1	1♀+2♂***
		2001			1	2	
Fetos	N	2000		1	3	1	1♀
		2001	1				1♂
Other places**		1999		2			
		2000		1			1♀

- Animals marked with radio-collars; ** - latrines and tracks; *** - One of the marked animals was initially captured when juvenile and recaptured, six months later, with an adult size, being then marked with a radio-collar.

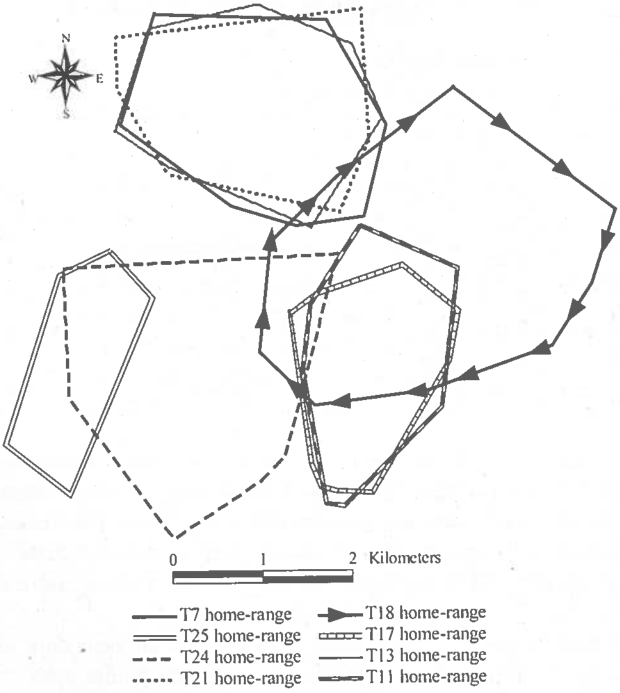

Figure 1. Badges home-ranges

radio-tagged animal, two of which have similar home-ranges that partially overlap with the one of the third; iii) the southwest group, with two marked badgers that have home-ranges partially overlapping. It is important to point out that in this last group the third animal (T18, an adult male) could be dispersing, a fact that could justify the low home-range overlap with the other two individuals. The data analysis of the southwest group, where only two animals are being monitored, should be regarded with cautioness because those animals were only recently radio-tagged (July and August 2001, respectively).

The existence of a sett with a high use rate (>90%) and fresh latrines, 2 km far north of the northern group home ranges, lead us to assume the presence of fourth badger clan in the 35 km^2 study area. Unfortunately, logistical constraints made impossible the implementation of a capture program in this northern sett.

In the cork oak woodland the mean home-range size was 4,92 km^2, being bigger for males (5. 98 km^2) than for females (4. 39 km^2) (Tab. 2). These values were similar to those published for southern Spain (REVILLA, 1998), but five to ten times lower than the ones referred for England (NEAL & CHEESEMAN, 1996), where the higher availability of resources (especially trophic) and the low interspecific competition with other carnivores for those resources allow an higher badger density and, consequently, smaller home-ranges.

Table 2. Badgers home-range size

Animal	Beginning of radio-tracking	Total number of locations	Home-range (km^2)
T7	09-05-2000	652	5. 13
T11	10-10-2000	313	3. 63
T13	13-06-2000	467	4. 87
T17	07-08-2000	397	3. 16
T18	16-10-2000	246	9. 02
T21	08-05-2001	123	4. 74
T24	30-06-2001	42	6. 51
T25	16-08-2001	32	2. 26

The high degree of overlapping in the intragroup home ranges and core-areas (Figs. 1 and 2), associated with a low inter-group overlap, suggests that this carnivore population, in this study area, maintains the social structure found in northern Europe with well-defined social units, although less dense. It is interesting to point out that the exclusive use of some areas is more evident in the core-areas, although there is a high intragroup overlap.

The combined analysis of the radio tracking and capture data allowed us to confirm the social behaviour of the Eurasian badger in this cork oak woodland area, of Southwest Portugal. Thus, its possible to state that, in this area, the social groups could be composed, at least, by three to four adult animals plus three to four juveniles, in a total of six to eight animals. With these values we can infer that badger density in our study area could vary between 0. 69 and 0. 91 badgers/km^2. This density is similar to the one referred by REVILLA (1998), for the Doñana Natural Park (South Spain). Inversely, our values are lower than those publish for some regions of England, where the densities could reach 25. 3 badgers/km^2 (ROGER *et al.,* 1997). This could be related to the density and regularity of their main prey in the area (earthworms) that possibly will not limit this predator population in the same way as the fluctuating trophic resources (insects and fruits) do in this Mediterranean habitat.

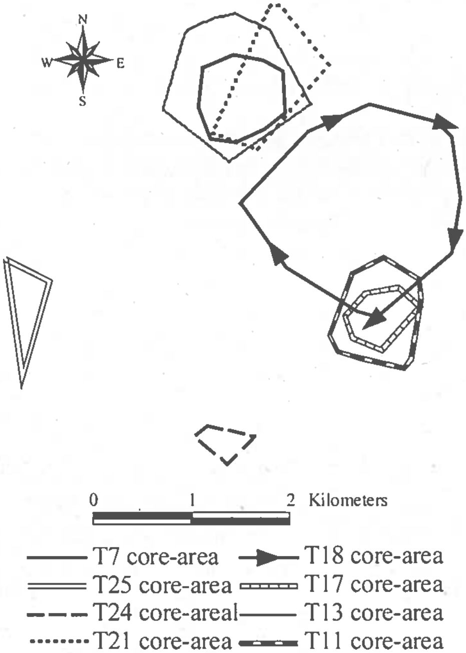

Figure 2 – Badgers core-areas

ACKNOWLEDGEMENTS

We are grateful for the financial support given by the Fundação para a Ciência e Tecnologia (PRAXIS XXI/BD/15842/98) and for the logistical support of the Centre of Environmental Biology. We would like to thank Maria João Santos, Maria João Cruz, Hugo Costa, Silvia Carvalho, Catarina Rei for the help in the field work.

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DISTRIBUTION AND ABUNDANCE OF THE LUSITANIAN TOADFISH *HALOBATRACHUS DIDACTYLUS* (BLOCH & SCHNEIDER, 1801) IN PORTUGAL WITH SOME REMARKS ON ITS POPULATION FRAGMENTATION

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Costa, J. L. & Costa, M. J. (2002). Distribution and abundance of the Lusitanian toadfish *Halobatrachus didactylus* (Bloch & Schneider, 1801) in Portugal with some remarks on its population fragmentation. *Revista Biol. (Lisboa)* **20**: 155-167.

The Lusitanian toadfish *Halobatrachus didactylus* (Bloch & Schneider, 1801) is a North-eastern Atlantic benthic fish occurring from the Bay of Biscay to the Gulf of Guinea. In Portugal it ranges mainly from Cabo Carvoeiro to the River Guadiana. In order to improve the knowledge about Lusitanian toadfish distribution and abundance on the Portuguese coast a total of 254 questionnaire surveys were carried out in 57 fishermen communities from Nazaré to V.R. S$^{\underline{to}}$ António. It was noted that the species is very abundant in the major estuaries (Tagus, Sado, Mira, Arade and Guadiana) and in the open coastal lagoons (Ria de Alvor and Ria Formosa) but absent from semi-closed coastal lagoons and littoral streams. Some notorious differences were also detected in the oceanic realm, with the Lusitanian toadfish being common in Algarve but much scarcer on the western coast of the country. Here, apart from the above estuaries this fish is reduced to minor marine nuclei. Since they seem to promote the reproduction and the recruitment of the species, high water temperatures, low hydrodynamic conditions and important freshwater flows apparently improve abundance of *H. didactylus*. Due to the strong wave action and especially to the low sea water temperature in the western Portuguese coast, it is not surprising that in this area the Lusitanian toadfish is mostly restricted to estuaries. In these systems the species finds adequate freshwater flows, good hydrodynamic conditions and high thermal levels during the reproduction period (late spring and early summer). This seems to be a case of population fragmentation induced by natural causes.

Key words: *Halobatrachus didactylus*, Portugal, distribution, abundance, population fragmentation

Costa, J. L. & Costa, M. J. (2002). Distribuição e abundância do xarroco, *Halobatrachus didactylus* (Bloch & Schneider, 1801) em Portugal com algumas observações sobre a sua fragmentação populacional. *Revista Biol. (Lisboa)* **20**: 155-167.

O xarroco *Halobatrachus didactylus* (Bloch & Schneider, 1801) é um peixe bentónico que se distribui no Atlântico nordeste entre os golfos da Biscaia e da Guiné. Os seus efectivos em Portugal continental repartem-se sobretudo entre o Cabo Carvoeiro e a foz do Guadiana. Para determinar com maior rigor a sua distribuição e abundância ao longo da costa lusitana foram efectuadas entrevistas às comunidades piscatórias situadas entre a Nazaré e V.R. S$^{\underline{to}}$ António. No total foram visitadas 57 localidades e realizados 254 inquéritos. Este trabalho revelou que a espécie é particularmente comum nos sistemas salobros em permanente comunicação com o mar a sul do Lis (estuários do Tejo, Sado, Mira, Arade e Guadiana e rias de Alvor e Formosa), estando ausente naqueles em que a ligação ao oceano é frequentemente interrompida (lagoas costeiras) e nas pequenas ribeiras que desaguam directamente no Atlântico. Em meio marinho o padrão de distribuição e abundância é completamente distinto nas costas meridional e ocidental. Embora menos comum que em ambiente salobro, no litoral oceânico algarvio o xarroco apresenta globalmente populações com alguma expressão. Pelo contrário, a norte do Cabo de S. Vicente apenas subsistem em meio marinho alguns núcleos populacionais de reduzida dimensão. Aparentemente, os quantitativos de *H. didactylus* são favorecidos por temperaturas elevadas, baixo hidrodinamismo e caudais dulçaquícolas relevantes, os quais parecem ter um papel determinante na reprodução e no recrutamento da espécie. Deste modo, o forte hidrodinamismo e, sobretudo, a reduzida temperatura da água do mar na costa ocidental portuguesa obrigam a que nesta região mais setentrional as populações de xarroco se encontrem praticamente confinadas aos meios salobros. Aqui existe água doce em abundância, a ondulação é residual e durante a Primavera e o Verão, quando a espécie se reproduz, a temperatura da água é francamente superior à verificada nas zonas marinhas adjacentes. Tudo indica que se trata, portanto, de um caso de fragmentação populacional devido a causas naturais.

Key words: *Halobatrachus didactylus*, Portugal, distribuição, abundância, distribuição fragmentacional.

INTRODUCTION

Like most batrachoidids, the Lusitanian toadfish *Halobatrachus didactylus* (BLOCH & SCHNEIDER, 1801) is a benthic species living partly buried on soft bottoms or concealed in rock crevices (ROUX, 1981). This relatively sedentary and solitary fish (MUZAVOR *et al*., 1993) occurs in shallow habitats down to about 50 m depth either in salt- or brackish water environments (SOBRAL & GOMES, 1997). It is a North-eastern Atlantic species usually considered distributed between the southern Bay of Biscay and the Gulf of Guinea, including the Western Mediterranean and some coastal islands (FOWLER, 1936; ROUX, 1986). However, in contrast to BAUCHOT (1987), the last author considers its occurrence merely occasional in the north of Spain and SOBRAL & GOMES (1997) go even further, considering *H. didactylus* rare or absent in the most northerly Portuguese estuaries. The lack of reliable data on the distribution and abundance of the species cause the discrepancies among these statements.

Thus, the aim of the present study was to determine the distribution and abundance of *H. didactylus* in Portugal, where the species is roughly at the northern limit of its geographic range. The influence of abiotic parameters on its distribution and abundance was also analysed.

METHODOLOGY

The distribution and abundance of *H. didactylus* in Portugal were determined by questionnaire surveys carried out in the centre and south of the country, between Nazaré and V.R. $S^{\underline{to}}$ António (fig. 1), since previous field operations together with the available literature (JORGE, 1991, REBELO, 1992, CUNHA & ANTUNES, 1993, FIDALGO & CORREIA, 1995) confirmed the absence of established populations of this species in the north of Portugal. During the last quarter of 1996 the results of a total of 254 interviews were obtained in 57 littoral settlements (maximum of five questionnaires per site). In order to improve the credibility of the collected data all interviews were made to groups of at least two fishermen. Questionnaires began with the exhibition of a Lusitanian toadfish photograph and they only proceeded if the fishermen recognised the species. In marine communities a single question about *H. didactylus* abundance was asked: in this place toadfish is (1) never caught; (2) rarely caught; (3) common but not abundant; (4) abundant; or (5) very abundant. In continental coastal systems a second question was asked: in this water body where is the location of (1) the upper and lower limits of distribution; and (2) the highest density. Other kinds of questions about the ecology of the species allowed ascertaining the truthfulness of the information supplied by the fishermen. All doubtful questionnaires were removed from the final sample.

Based on the answers to the first of the two questions raised in the interviews a semi-quantitative abundance index was computed:

$$\mathrm{SQAI} = \frac{P_1 + P_2 + P_3 + \ldots + P_n}{n};$$

where P_1, P_2, P_3 and P_n are the values attributed per questionnaire (never caught - 0; rarely caught - 1; common but not abundant - 2; abundant - 3; very abundant - 4); and n the total number of questionnaires considered in each case. This index allows the classification of different areas according to the species abundance:

$SQAI=0$ ➤ Absent
$0<SQAI\leq1$ ➤ Occasional
$1<SQAI\leq2$ ➤ Little abundant
$2<SQAI\leq3$ ➤ Abundant
$3<SQAI\leq4$ ➤ Very abundant

For the open sea, this index was computed separately for each location ($SQAI_l$) and in an aggregate form for confluent zones with approximate levels of abundance ($SQAI_z$), in order to obtain a global quantitative picture of the species distribution along the Portuguese coast. For continental coastal systems only $SQAI_z$ values were computed, with internal distribution and abundance variability determined from the information gathered from the second question.

The influence of some environmental factors on species abundance in the sea was also investigated by means of a multiple linear regression model relating $SQAI_l$ (dependent variable) with the following site variables (independent variables): (1) surface water temperature, obtained from MOURA *et al.* (1988); (2) surface water salinity, obtained from MARTINS (1997); (3) least distance to brackish water toadfish populations, used in this analysis to verify the influence of these continental nuclei on the open sea species abundance due to the migration of some individuals; (4) hydrodynamic conditions, represented by the number of days per year with waves higher than 1 m, and determined according to PIRES (1989); (5) presence of geographic features protecting the coastline from NW undulation, the dominant wave direction (about 80% of the year) in Portuguese western coast (PIRES, 1989); (6) prevalence of rocky substratum (more than 50% of the bottom covered with rocks within a 5 km extension for each side and up to a depth of 50 m), according to FRANCA & COSTA (1984), INIP & IH (1988) and IPIMAR & IH (1996, 1999); and (7) proximity to important freshwater sources (presence of rivers with more than 100 km^2 of catchment area in the vicinity of 5 km), according to ANONYMOUS (1981).

Because water temperature and undulation are highly correlated in the Portuguese coast ($r=-0.96$; $df=38$; $p<0.001$), these variables could not be used simultaneously in multiple regression analysis due to problems of multicollinearity (HAIR *et al.*, 1998). Therefore, they were replaced by Latitude, which is significantly correlated with both temperature ($r=-0.74$; $df=38$; $p<0.001$) and undulation ($r=0.84$; $df=38$; $p<0.001$). This regression analysis was performed by the stepwise selection method (probability-of-F-to-enter≤0.05 and probability-of-F-to-remove≥0.10) using the SPSS statistical package.

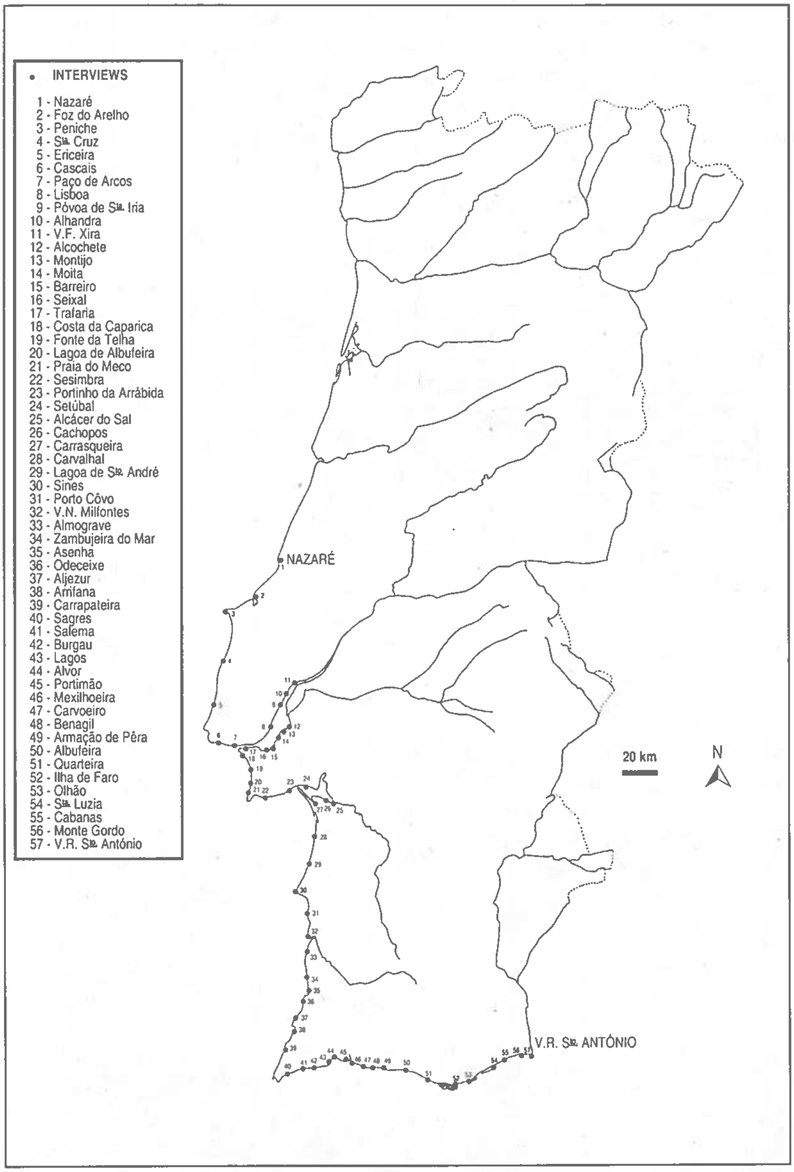

Figure 1. Portuguese coastal settlements where questionnaire surveys were conducted

RESULTS AND DISCUSSION

The results obtained in this work together with some bibliographic sources (JORGE, 1991, REBELO, 1992, CUNHA & ANTUNES, 1993, FIDALGO &

CORREIA, 1995) indicate that populations of *H. didactylus* are absent from the northern coast of the country (fig. 2), although from time to time some specimens can appear in this area (ROUX, 1986, BAUCHOT, 1987).

In southern continental coastal systems the species is very abundant only in estuaries (Tagus, Sado, Mira, Arade and Guadiana) and open lagoons (Alvor and

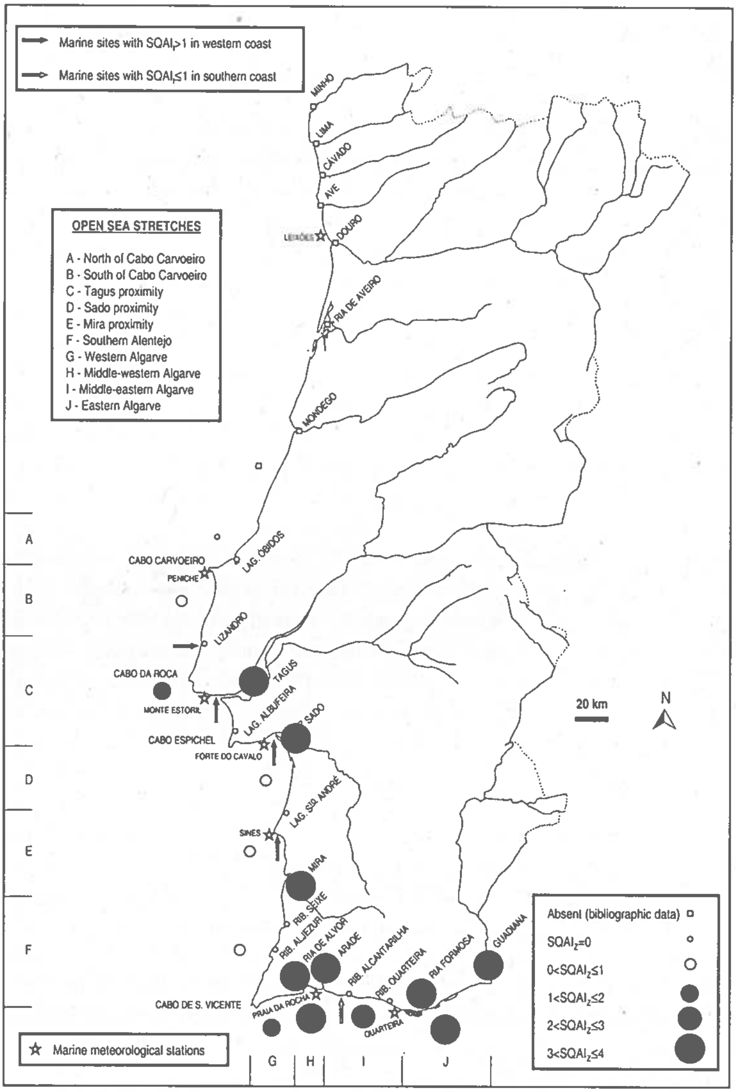

Figure 2. Distribution and abundance of *H. didactylus* in Portugal

Formosa), being completely absent from all studied semi-closed lagoons (Óbidos, Albufeira and S$^{\underline{to}}$ André) and small littoral rivers (Lizandro, Seixe, Aljezur, Alcantarilha and Quarteira). This pattern of distribution is highly consistent with earlier scientific works, since the species was previously reported from the Tagus (COSTA *et al.*, 2000), Sado (CUNHA, 1994), Mira (COSTA *et al.*, 1987) and Guadiana (SANTOS *et al.*, 2000) estuaries, and from the Alvor (ANTUNES & CUNHA, 1986) and Formosa (MONTEIRO, 1989) lagoons, but never from the Seixe or Aljezur rivers (ALVES, 1988) or the Óbidos (COSTA *et al.*, 1994), Albufeira (ANTUNES & CUNHA, 1993) or S$^{\underline{to}}$ André (BERNARDO, 1990) lagoons. Such differences are most likely a result from the ecological characteristics of this species, which is relatively sedentary in the adult stage (MUZAVOR *et al.*, 1993), exhibits low fecundity (ALMEIDA *et al.*, 1997), benthic eggs and larvae, and displays parental care with the offspring (MAIGRET & LY, 1986). Thus, although highly adapted to dominating habitats with already established populations, *H. didactylus* shows some difficulties in terms of dispersal and colonisation of new territories. Therefore, in very unstable environments, such as semi-closed lagoons and coastal rivers, the frequency of catastrophic events is expected to be greater than the rate of colonisation by the species, resulting in its absence from such water bodies. Conversely, in more stable environments, like open lagoons, estuaries and the sea, the species is usually present along its geographic range, occasionally in great abundance.

According to the answers obtained for the second question asked regarding continental coastal systems, the species is well established in the middle reaches of estuaries, being much scarcer upstream. On the contrary, in open lagoons *H. didactylus* does not show such a spatial pattern of abundance, which can be associated with the influence of salinity. In fact, salinity is fairly constant throughout open lagoons (ANTUNES & CUNHA, 1986, MONTEIRO, 1989) but in estuarine systems it shows a clear decrease in the upstream direction (COSTA *et al.*, 1987, 1996, CUNHA, 1994). A downstream smooth decreasing trend in species abundance is also detected in south-western estuaries (Tagus, Sado and Mira).

The abundance pattern of this species in open sea is very different from the one observed in continental coastal systems (fig. 2). In the ocean the abundance of *H. didactylus* is only important in the Central and Eastern Algarve (mainly in the vicinity of Guadiana/Ria Formosa and Arade/Ria de Alvor systems), being much lower in regions under more Atlantic influence such as Western Algarve and especially off the western coast. As stated before, no populations of this species seem to be established north to Cabo Carvoeiro. Between this geographic feature and Cabo de S. Vicente only some minor marine nuclei of Lusitanian toadfish were detected. They are located close to the mouth of River Lizandro and south from Cabo da Roca (the most important one), Cabo Espichel and Cabo de Sines (fig. 2). Apart from these small populations only isolated individuals of

H. didactylus can be expected off the west coast. Once again, a revision of the available literature confirmed this pattern of distribution and abundance in the open sea, since captures of the species were only obtained, respectively by GONÇALVES (1941), HENRIQUES *et al.* (1999), HELLING (1943) and SANTOS (1997), at Cascais (south to Cabo Raso), Arrábida (south to Cabo Espichel), Sines (south to Cabo de Sines) and Faro/Olhão (Eastern Algarve). In contrast to Arrábida, where the species is not abundant (HENRIQUES *et al.*, 1999), marine populations from Algarve are more important (SANTOS, 1997). The best model relating $SQAI_l$ in the sea with the chosen abiotic parameters explained more than 60% of the variability ($r^2_{adj}=0.61$; F=21.55; df=2, 36; $p<0.001$). The three predictors included in the model were Latitude, NW protection, and the presence of freshwater sources (tab. 1). The unstandardized coefficients of regression (*B* ± standard error) revealed that the Lusitanian toadfish abundance increases at lower latitudes and in areas protected from NW undulation receiving important freshwater discharges. According to the *t*-test values obtained in order to verify the relative importance of each variable in the model (ANONYMOUS, 1997), the presence of freshwater flows is the least relevant of these parameters.

Table 1. Best multiple linear regression model describing the relationship between some abiotic parameters and *H. didactylus* abundance in the open sea

VARIABLES	B ± SE	t	P
Latitude	- 0.63 ± 0.14	- 4.56	< 0.001
NW protection	0.96 ± 0.23	4.21	<0.001
Freshwater sources	0.73 ± 0.22	3.26	<0.01
Constant	4.52 ± 5.30	4.62	< 0.001

The increase in the abundance of *H. didactylus* in the vicinity of freshwater discharges is probably related to the high productivity of these areas due to the transport from the continent of large amounts of nutrients and organic matter. This hypothesis is supported by the fact that the most successful recruitment of the species occurs in very rainy years, especially when the rain is concentrated in the winter season, corresponding to the pre-reproductive period (unpublished data). Thus, important freshwater flows may improve food availability during sexual maturation and early development of the offspring, contributing to the species recruitment success and its global abundance.

Protection from NW undulation is clearly a very relevant factor for the establishment of important *H. didactylus* populations, since all but one open sea location (Ericeira) with $SQAI_l>1$ are not exposed (fig. 2). Strong hydrodynamic conditions may turn adult movements and normal activity difficult, and increase the rates of mortality both of eggs and larvae due to their benthic nature.

The slight global increase in species abundance near Tagus mouth could be related with the important freshwater flows discharged by this river (the most relevant of the South-western Iberian Peninsula) as well as with the protection from the NW undulation provided by the Cabo da Roca geographic feature. Important freshwater flows and residual wave action probably also strongly influence the abundance of *H. didactylus* in brackish water environments.

The relative importance of temperature and undulation on *H. didactylus* abundance along the Portuguese coast is impossible to determine because both variables are highly correlated in this region. However, hydrodynamic conditions could not explain the absence of Lusitanian toadfish from the northerly Portuguese estuaries, where the wave action is only residual. Therefore, temperature may be considered the limiting factor for the occurrence of *H. didactylus* established populations.

Along the Portuguese coast this species spawns in late spring and early summer when water temperature increases to more than 17-20 ºC (unpublished data), just like the North-western Atlantic batrachoidids of the genus *Opsanus* (BREDER, 1941, WALTERS & ROBINS, 1961, LOWE, 1975). Thus, in open Portuguese waters, the temperature required for Lusitanian toadfish repopulation is only commonly attained in the Algarve, in contrast to brackish water systems where water temperatures rise up to more than 20 ºC in late spring and early summer, even on the south-west coast (tab. 2).

Table 2. Mean water and air temperature (ºC) in different places along the Portuguese coast from May to August. Information obtained from: (1) MOURA *et al.* (1988), (2) COSTA *et al.* (1996), (3) SOBRAL (1981) and (4) COSTA *et al.* (1987)

WATER TEMPERATURE (sea) (1)	MAY	JUN	JUL	AUG
Leixões	14.3	15.0	15.4	15.3
Peniche	14.8	15.6	16.3	16.6
Monte Estoril	15.3	16.3	16.7	16.5
Forte do Cavalo	15.2	16.4	16.8	17.3
Sines	14.6	15.4	16.2	16.1
Praia da Rocha	16.8	18.6	20.1	20.2
Quarteira	17.8	19.0	20.7	20.9
WATER TEMPERATURE (estuaries)	**MAY**	**JUN**	**JUL**	**AUG**
Tagus (central area) (2)	16.8	21.5	22.8	21.3
Sado (lower reaches) (3)	17.8	19.4	22.4	21.9
Mira (all water body) (4)	16.4	22.7	23.4	22.4
AIR TEMPERATURE (estuaries/lagoons) (1)	**MAY**	**JUN**	**JUL**	**AUG**
Lima (Viana do Castelo)	15.4	18.1	19.8	18.9
Ria de Aveiro (S. Jacinto)	15.6	17.3	18.3	18.3
Mondego (Barra do Mondego)	16.2	18.0	18.9	19.1
Tagus (Lisbon)	17.7	20.6	22.4	22.7
Sado (Setúbal)	17.5	20.4	22.3	22.6
Arade (Praia da Rocha)	17.6	20.6	22.8	23.1
Ria Formosa (Faro)	18.6	21.6	23.9	24.1
Guadiana (V.R. Sto. António)	18.4	21.4	23.6	23.6

Brackish water thermal conditions depend on the sea and air temperatures, and despite the lack of reliable data series, it is highly probable that major discontinuities in this parameter occur between Mondego and Tagus due to a pronounced increase in air temperature from north to south in this area (tab. 2).

Therefore, thermal requirements for reproduction of *H. didactylus* are satisfied both in the Algarve marine and brackish water environments, with the species being fairly abundant throughout this region. In the south-west coast (between Cabo Carvoeiro and Cabo de S. Vicente) it is only in the estuaries that water temperatures reach more than 17 ºC with some regularity, enabling the establishment of important populations in these systems. In the open sea only some minor population nuclei are present in the most protected areas with a few isolated animals also found in this region. North to Cabo Carvoeiro both water and air temperatures continue to decrease and reproduction is impossible even in brackish water systems. Thus, in this northern region only some wandering individuals can be found. These thermal differences could also explain why the reduction of *H. didactylus* abundance in estuarine lower reaches is more pronounced on western coast than in the south.

Although less determinant than the other three referred factors, the presence of brackish water populations should also be considered relevant to the abundance of this species in the adjacent marine areas. In fact, only the migration of some individuals from the Arade estuary and Ria Formosa to oceanic waters could justify the observed pattern of abundance between these two systems (fig. 3), since no important variations occur here with regards to temperature, undulation or freshwater inputs - affluence of freshwater to coastal ecosystems is usually reduced in Algarve except near Guadiana river (COSTA *et al*., 1985).

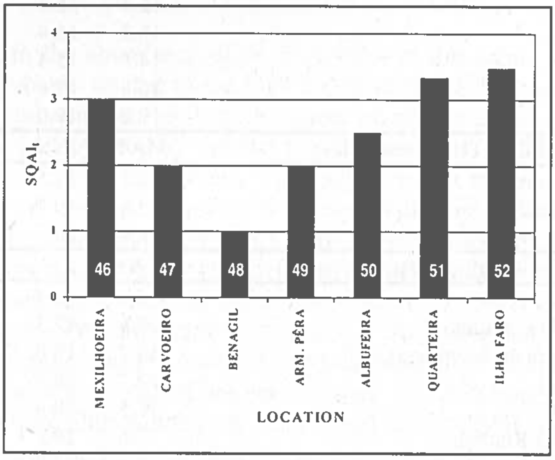

Figure 3. Abundance of *H. didactylus* in the coastal area between the Arade estuary and Ria Formosa (reference numbers of coastal settlements according to fig. 1)

Even in the western coast, in more protected areas (south to Cabo da Roca and Cabo Espichel), the presence of brackish water Lusitanian toadfish populations could be partially responsible for an increase in adjacent marine numbers. However, the differences between western and southern coasts were responsible for the exclusion of this variable from the final model.

At present, Portuguese populations of this subtropical species seem to be fragmented due to a set of natural causes. The degree of isolation between these populations and its consequences should be studied in the future through morphometric, meristic, parasitological and genetical analysis. Implications of global warming on species distribution and abundance should also be investigated.

ACKNOWLEDGEMENTS

Our gratitude is expressed to I. Domingos for critically reviewing the manuscript. Two anonymous referees are also thanked. Financial assistance was provided by the Fundação para a Ciência e a Tecnologia.

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EFEITO DO TIPO DE FLORESTA NO PADRÃO DE DISTRIBUIÇÃO DE *ARMERIA ROUYANA* DAVEAU NA COMPORTA/GALÉ E NO ESTUÁRIO DO SADO

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Clemente, A. S., Chozas, S., Repas, M. & Martins-Loução, M. A. (2002). Efeito do tipo de floresta no padrão de distribuição de *Armeria rouyana* Daveau na Comporta/Galé e no Estuário do Sado. *Revista Biol. (Lisboa)* **20**: 169-177.

Armeria rouyana Daveau é uma planta endémica do SW de Portugal, característica de matos e pinhais abertos sub-litorais de dunas fixas. Esta espécie ocorre em espaços abertos, com alguma perturbação e, embora seja uma espécie prioritária do Anexo II da Directiva Habitats (92/43/EEC), o conhecimento da sua ecologia e distribuição é ainda escasso. Uma das principais áreas de distribuição de *A. rouyana* localiza-se nas Zonas Especiais de Conservação Comporta/Galé e Estuário do Sado, caracterizadas por extensas explorações florestais de pinheiro e eucalipto. As florestas tradicionais estão a ser progressivamente substituídas por práticas florestais mais intensivas, que poderão afectar a abundância local da espécie e conduzir à fragmentação do seu habitat. Neste trabalho, foi comparada a abundância de *A. rouyana* em florestas com idade e densidade diferentes na Comporta/Galé e no Estuário do Sado. A espécie registou maior abundância, com núcleos populacionais de grandes dimensões, em situações de matos baixos e esparsos e em pinhais jovens. As florestas de eucalipto e de pinhal adulto, com sub-bosque de mato alto e denso, apresentaram uma abundância muito inferior, com pequenos núcleos da espécie. Os resultados indicam que a intensificação da exploração das florestas poderá conduzir à fragmentação do habitat e à redução das populações de *A. rouyana*. No presente trabalho são propostas medidas de gestão destinadas a contrariar a actual tendência e a valorizar os principais núcleos populacionais, através da manutenção e implementação de práticas silvícolas adequadas à conservação da espécie.

Palavras-chave: conservação, endemismo, gestão, uso da floresta.

Clemente, A. S., Chozas, S., Repas, M. & Martins-Loução, M. A. (2002). Effect of forest management on the distribution pattern of *Armeria rouyana* Daveau in Comporta/Galé and Estuário do Sado. *Revista Biol. (Lisboa)* **20**: 169-177.

Armeria rouyana Daveau is an endemic plant species from SW Portugal, occurring in open pinewoods of inland dunes. This species tends to be more frequent in disturbed forest gaps or clearings but, despite being a priority species under the Habitats Directive (92/43/EEC), the information available concerning its ecology and distribution is still scarce. The Special Areas of Conservation Comporta/Galé and Estuário do Sado, where extended forests of pine and eucalyptus are exploited, represent most of the distribution range of *A. rouyana*. Since traditional forest exploitation has been replaced by more intensive forest practices, local abundance of this species is expected to be affected, as well as habitat connectivity. In this work we aimed to compare the abundance of *A. rouyana* in pine and eucalyptus forests of Comporta/Galé and Estuário do Sado differing in density and age. The highest abundance was recorded in sparse shrublands associated with open pinewoods and in young pine plantations. Mature euclyptus and pine forests, with a dense understory, recorded the lowest abundance levels, with a few plants sparsely distributed. Our results indicate that the present exploitation of dense forests may lead to reductions in population numbers and habitat fragmentation. Strategies for conservation of populations based on sustainable forest management are suggested.

Keywords: conservation, endemism, forest management.

INTRODUÇÃO

A distribuição de uma espécie no tempo e no espaço pode diferir da sua distribuição potencial determinada pelas condições edafo-climáticas. Os factores de perturbação influenciam a distribuição das espécies, alterando directamente a disponibilidade de recursos, criando oportunidades para o estabelecimento de novas espécies ou reduzindo as populações das espécies existentes (e.g. HOBBS & HUENNEKE, 1992, VAN DER MAAREL, 1993, MOTZKIN *et al.*, 1999). Nos ecossistemas mediterrânicos, estes efeitos são particularmente notórios, dada a longa história de actividade humana associada à floresta (BLONDEL & ARONSON, 1999).

Armeria rouyana Daveau (Plumbaginaceae) é uma espécie endémica do Noroeste Alentejano e da bacia inferior do Tejo, característica de matos e pinhais sub-litorais de dunas fixas. Planta vivaz, de pequeno porte, ocorre em clareiras de matos associadas a alguma perturbação (PINTO, 1996). Embora seja uma espécie prioritária do Anexo II da Directiva Habitats (92/43/EC), o conhecimento da sua ecologia é ainda muito escasso.

As florestas de pinheiro e eucalipto constituem a principal ocupação do solo numa das principais áreas de distribuição de *A. rouyana*: as Zonas Especiais de Conservação (ZECs) Comporta/Galé e Estuário do Sado, criadas no âmbito da Rede Natura 2000. A implementação de práticas silvícolas intensivas e a expansão destas florestas poderá afectar a distribuição da espécie, quer pelo efeito directo do ensombramento, quer pela fragmentação do habitat. Esta última pode conduzir à extinção de algumas populações e/ou criar barreiras à dispersão (DEBINSKI & HOLT 2000). Embora os requisitos ecológicos da espécie indiquem uma preferência por locais abertos, os efeitos do tipo de floresta na sua abundância são ainda desconhecidos. Este trabalho teve como objectivo comparar a abundância de *A. rouyana* em florestas com diferente idade e densidade nas ZECs Comporta/Galé Estuário do Sado.

ÁREA DE ESTUDO

O trabalho foi realizado na área abrangida pelas ZECs (Zona Especial de Conservação) Comporta/Galé e Estuário do Sado (UTM 510–540 E, 4235–4275 N) (Fig. 1). Nesta região, as florestas de pinheiro e eucalipto constituem a principal ocupação do solo (Fig. 2).

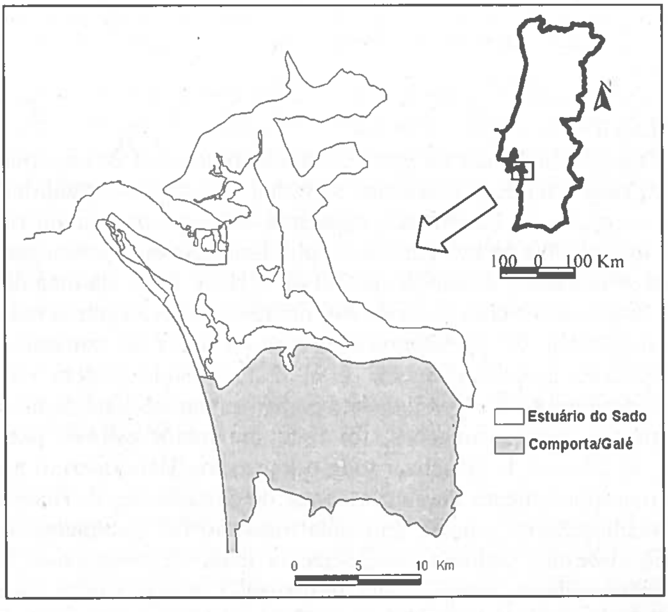

Figura 1. Localização das áreas de estudo (Zona Especial de Conservação Estuário do Sado e Zona Especial de Conservação da Comporta/Galé).

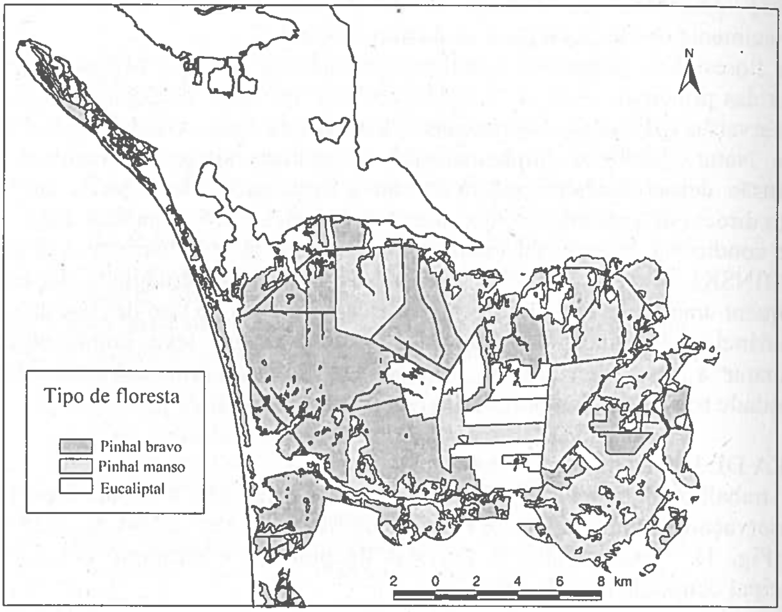

Figura 2. Distribuição das florestas de pinheiro e eucalipto na área de estudo (adaptado de AFLOPS 2000).

METODOLOGIA

A amostragem cobriu toda a área de estudo, num total de 28 quadrículas UTM 5x5Km. Em cada uma destas quadrículas foram amostradas 5 quadrículas UTM 1x1Km, à excepção de quadrículas com área coberta por mar ou rio, onde o número de quadrículas de 1x1 Km foi proporcional à área ocupada por terra. No total, foram amostradas 74 quadrículas UTM 1x1Km. Em cada uma delas foram realizados 16 transectos com cerca de 140 m, de direcção aleatória, registando-se o número de plantas de *A. rouyana* observado a partir do transecto, em duas faixas laterais de aproximadamente 2 m. Este método poderá ter um erro associado à diminuição da visibilidade das plantas em núcleos de mato denso e alto, no entanto, nestas situações, foi feito um maior esforço por parte do observador no sentido de visualizar todo o transecto. Além disso, o trabalho foi efectuado maioritariamente durante a época de floração de *A. rouyana*, o que facilita a localização da espécie. Em cada transecto foi igualmente registado o tipo de uso florestal, incluindo a espécie, a idade aproximada e as práticas silvícolas associadas (vestígios de desmatação ou desbaste recente, por exemplo). Para este trabalho, foram seleccionados apenas os transectos correspondentes aos quatro tipos de floresta mais frequente: Pinhal aberto com

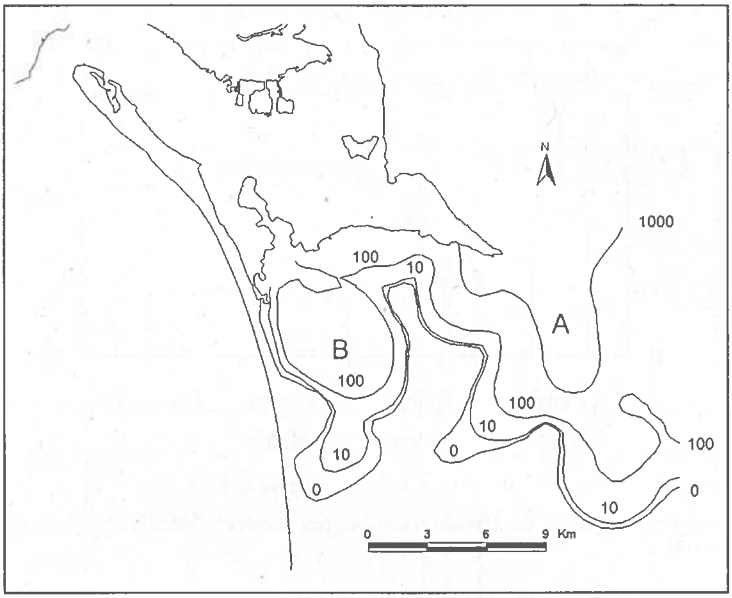

Figura 3. Distribuição de *Armeria rouyana* na área de estudo. As isolinhas definem áreas de igual abundância da espécie. A e B representam os dois núcleos principais da espécie.

mato (pinhal de baixa densidade com sub-bosque de matos), Pinhal jovem (plantações de pinheiro em linha com 10-20 anos), Pinhal adulto (plantações de pinheiro com 30-40 anos, normalmente com sub-bosque de matos altos e densos) e Eucaliptal (plantações de eucalipto adulto). Foram realizados 80, 80, 128 e 128 transectos em cada tipo de floresta, respectivamente.

Foi gerado um mapa de distribuição de *A. rouyana* para a área de estudo (CEBV 2001), com base na estimação por "kriging" e utilizando o programa Surfer 6. A associação entre a abundância de *A. rouyana* e o tipo de floresta foi testada com uma ANOVA de um factor, seguida do teste de comparação múltipla de Tukey no programa Statistica. No sentido de obter a homogeneidade de variâncias e distribuição normal dos dados, foi utilizada a transformação de raiz quadrada (ZAR, 1984).

RESULTADOS

Armeria rouyana apresenta dois núcleos principais na zona de fronteira entre as duas ZECs, no sector sul do Estuário do Sado e sector norte da Comporta/Galé (Fig. 3). Estes dois núcleos estão em contacto apenas na faixa norte, existindo uma descontinuidade a sul. Esta descontinuidade coincide com as áreas de ocorrência de florestas de pinheiro manso e eucalipto de grandes dimensões (Fig. 2).

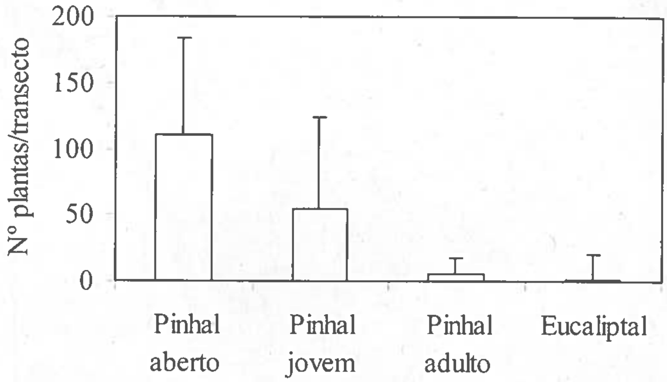

Figura 4. Número de plantas de *Armeria rouyana* por transecto (média±d.p.) nos quatro tipos de floresta.

Os quatro tipos de floresta estudados apresentaram diferenças significativas no número de plantas de *A. rouyana* (ANOVA, F=182, P<0,001) registando-se diferenças entre todos os tipos excepto entre pinhal adulto e eucaliptal (Fig. 4). Estes últimos apresentaram o menor número médio de plantas por transecto: apenas 5 e 2 plantas por transecto, respectivamente (Fig. 4).

A. rouyana registou abundância máxima em matos baixos e esparsos, associados a pinhal aberto (Fig. 5 A). Os pinhais jovens apresentaram uma abundância inferior à destas florestas, embora a proporção de transectos com 11-100 e >101 plantas ainda seja elevada (Fig. 5 B). Nos pinhais jovens sujeitos a desmatações regulares entre as linhas de pinheiros a presença das plantas de *A. rouyana* restringe-se aos matos residuais próximos das árvores e às margens dos aceiros e caminhos. Os pinhais adultos e eucaliptais, normalmente com copado denso e sub-bosque de mato alto e denso, apresentaram apenas alguns núcleos dispersos, normalmente com poucos indivíduos (Figs. 5 C e D). Em ambos, as plantas observadas localizaram-se em clareiras de matos e margens de caminhos. Na maioria dos casos, estes núcleos aparecem apenas na proximidade de outros tipos de floresta, onde a abundância da espécie é mais elevada.

DISCUSSÃO

Os resultados deste trabalho indicam que a abundância de *A. rouyana* diminui com a idade e a densidade da floresta (Fig. 5). Este padrão de abundância poderá estar relacionado com as características do habitat associado aos diferentes tipos de floresta. As florestas jovens e/ou com árvores dispersas apresentam clareiras onde esta espécie se estabelece, enquanto nas florestas adultas, a sua baixa densidade poderá estar associada ao ensombramento resultante da coalescência das copas. O ensombramento e a competição das espécies arbustivas presentes no sub-bosque poderão constituir os principais factores limitantes da presença de *A. rouyana*, tal

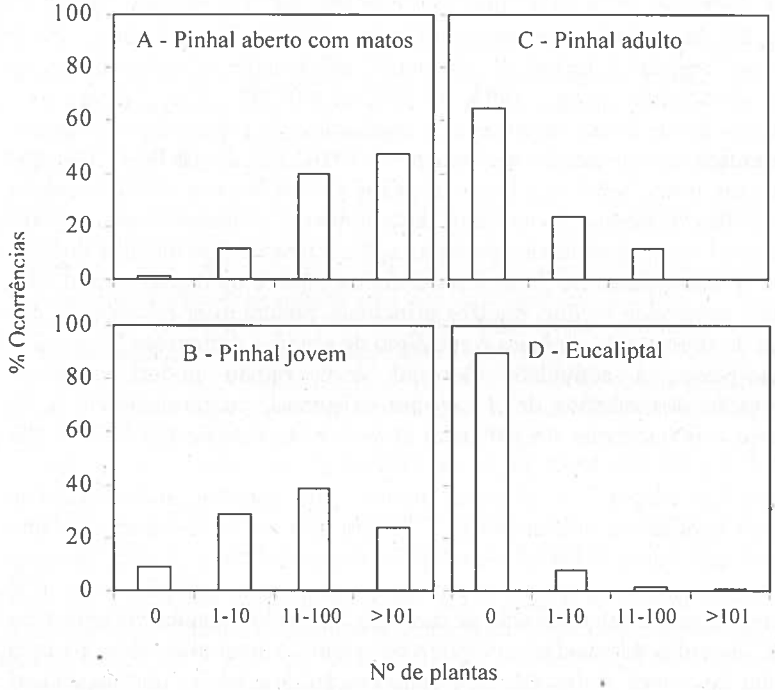

Figura 5. Percentagem de ocorrência das classes de abundância de *Armeria rouyana* nos quatro tipos de floresta.

como acontece com outras espécies de sub-bosque (MOTZKIN *et al.*, 1999).

Os ecossistemas naturais consistem em mosaicos de habitats onde diversos tipos de perturbações, como fogos ou períodos de seca, atrasam o processo de sucessão natural. As actividades associadas à exploração de recursos florestais poderão igualmente favorecer os estádios iniciais da sucessão, já que as actividades como desbaste ou desmatações e manutenção de caminhos e aceiros criam espaços abertos potencialmente colonizáveis por *A. rouyana* e outras espécies dependentes de perturbações. No entanto, *A. rouyana* não foi observada em diversas florestas adultas com clareiras, o que indica a necessidade de considerar a influência de outros factores na distribuição da espécie, tais como a sua capacidade de colonização, a frequência das perturbações e a área ocupada por cada tipo de floresta.

Actualmente, a preparação do terreno associada à sementeira ou plantação de pinhais e eucaliptos recorre a desmatações em grande escala, que eliminam localmente as espécies constituintes dos matos. Na maior parte dos casos, essas desmatações são frequentes durante os primeiros dez anos. Embora muitas das espécies sobrevivam a este tipo de perturbação, regenerando através da germinação de sementes presentes no solo, desmatações frequentes poderão reduzir ou esgotar o banco de sementes, eliminando definitivamente essas espécies (CANCIO *et al.*, 1993, LUIS-CALABUIG *et al.*, 1996). A re-colonização desses locais dependerá da capacidade de regeneração da espécie e da capacidade de dispersão de sementes (EHRLÉN & ERIKSSON, 2000), podendo ser muito lenta em locais isolados (SOULÉ *et al.*, 1992). Assim, a dimensão das florestas poderá ser determinante, sobretudo se as florestas tradicionais forem substituídas por explorações intensivas, de maiores dimensões e estrutura homogénea. De facto, a ausência da espécie na região central da área de estudo, separando os dois núcleos principais, poderá estar relacionada com a presença de florestas de pinheiro e eucalipto de grandes dimensões (Figs.s 2 e 3). A longo-prazo, a actividade florestal desta região poderá conduzir à fragmentação dos núcleos de *A.rouyana* existentes, comprometendo o fluxo genético e a manutenção do potencial evolutivo da espécie (ELLSTRAND & ELAM, 1993, NEWMAN & PILSON, 1997).

A evolução natural das florestas jovens para estádios sucessionais mais avançados condiciona a abundância e distribuição de *A. rouyana*, que deverá apresentar uma variação espacial e temporal dependente da estrutura, dimensão e idade do mosaico de florestas presente. Assim, a conservação desta espécie prioritária exige a implementação de medidas de gestão baseadas na aplicação de práticas silvícolas adequadas, que poderão constituir uma alternativa na criação de habitat favorável. Actualmente, a conservação de espécies que dependem de perturbações ou de habitats que tendem a desaparecer em estádios sucessionais mais avançados exige frequentemente a intervenção humana (SUTHERLAND 2000). Da mesma forma, a gestão do habitat de *A. rouyana* deverá envolver a reprodução das características naturais do habitat, mantendo artificialmente os estádios sucessionais iniciais. As medidas a implementar deverão regulamentar a frequência das desmatações, criando um limite mínimo de 3 a 4 anos, e visar a criação de uma rede de clareiras, de modo a manter a continuidade entre as populações da espécie. A proximidade de fontes de propágulos e a dimensão e complexidade estrutural das explorações florestais existentes ou a implementar deverão igualmente ser consideradas no planeamento das medidas de gestão.

AGRADECIMENTOS

Este trabalho foi realizado no âmbito do projecto LIFE NATUREZA "Rede Natura 2000 na Península de Setúbal/Sado" (B4-3200/98/400), coordenado pela Associação de Proprietários Florestais do distrito de Setúbal (AFLOPS).

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INVENTARIAÇÃO PRELIMINAR DE COMUNIDADES MACROFÚNGICAS EM MATAS DA SERRA DE SINTRA

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Branco, S. M. (2002). Inventariação preliminar de comunidades macrofúngicas em matas da Serra de Sintra. *Revista Biol. (Lisboa)* **20**: 179-190.

No presente trabalho inventariaram-se as comunidades macrofúngicas de um acacial, de um cupressal, de uma mata mista e de um pinhal situados na Serra de Sintra. O estudo decorreu entre Outubro de 2000 e Fevereiro de 2001. Na totalidade foram encontradas 139 espécies, sendo o acacial a mata com maior diversidade específica e o pinhal com a menor. As quatro matas mostraram micota bastante diferentes. As espécies saprotróficas dominaram no acacial, no cupressal e na mata mista, enquanto as espécies micorrízicas predominaram no pinhal. Os estudos sobre comunidades macrofúngicas devem prolongar-se entre três e cinco anos para produzirem informações consistentes. Assim, este trabalho deve ser encarado como preliminar e a sua continuação é imprescindível para um melhor conhecimento destas comunidades.

Palavras chave: Macrofungos, comunidades macrofúngicas, Serra de Sintra.

Branco, S. M. (2002). Preliminary inventory of macrofungal communities of Serra de Sintra's woods. *Revista Biol. (Lisboa)* **20**: 179-190.

In the present study macrofungal communities from an *Acacia longifolia* stand, a *Cupressus lusitanica* stand, a mixed wood stand and a *Pinus pinaster* stand were inventoried. The study took place between October 2000 and February 2001.

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139 species were recorded altogether. The *Acacia longifolia* stand had the highest species diversity and the *Pinus pinaster* stand had the lowest. All the four stands showed very different mycota. Saprotrophic species dominated in the *Acacia longifolia* stand, the *Cupressus lusitanica* stand and the mixed wood stand, while the mycorrhizal species dominated in the *Pinus pinaster* stand.

Macrofungal community studies should last three to six years to produce consistent information, therefore the present study should be seen as preliminary and should continue, in order to provide more information about these communities.

Keywords: Macrofungi, macrofungal communities, Serra de Sintra.

INTRODUÇÃO

Os macrofungos são os fungos que formam estruturas reprodutoras visíveis a olho nu (ARNOLDS, 1981). O termo comunidade macrofúngica, usado em analogia com ao termo comunidade vegetal, designa um conjunto concreto de fungos que cresce num determinado espaço uniforme, independentemente do seu tamanho e grau de heterogeneidade em termos da exploração de habitat e de preferências de substrato (ARNOLDS, 1992).

As comunidades macrofúngicas desempenham um papel fulcral no funcionamento dos ecossistemas florestais. Os macrofungos são considerados como bioindicadores capazes de informar sobre a área e o habitat onde surgem, podendo a inventariação de espécies macrofúngicas ser reveladora do estado dos ecossistemas (FELLNER e PEŠKOVÁ, 1995; STASIŃSKA, 1999; LAGANÀ *et al*, 2000).

O estudo das comunidades macrofúngicas é muito dificultado pela natureza dos macrofungos. O carácter efémero dos carpóforos (normalmente até cinco dias, segundo RICHARDSON, 1970), a irregularidade da frutificação das espécies, que leva a diferenças na diversidade de espécies nas mesmas comunidades de ano para ano (MILLER, 1995) e a impossibilidade de contar o número de indivíduos genética ou fisiologicamente distintos (SCHMIT *et al*, 1999) contribuem para a falta de conhecimentos na área.

Com a sugestão de que se assiste a um declínio da flora macrofúngica (ARNOLDS, 1988), a inventariação e o estudo dos macrofungos, mais do que importante, torna-se urgente.

No nosso país poucos investigadores se dedicaram ao estudo dos macrofungos e das comunidades macrofúngicas. Na Serra de Sintra existem apenas registos esporádicos de espécies macrofúngicas, não existindo até agora nenhum inventário de espécies.

O presente trabalho teve como objectivo o estudo de comunidades macrofúngicas em matas da Serra de Sintra. Procedeu-se à inventariação das espécies presentes num acacial, num cupressal, numa mata mista e num pinhal.

ZONAS DE ESTUDO

A Serra de Sintra insere-se no Parque Natural de Sintra – Cascais (PNSC), situado a Norte de Cascais (fig. 1). A Serra de Sintra possui condições climáticas muito particulares, com influências tanto atlânticas como mediterrânicas, e contrastantes com a aridez da zona envolvente. Apesar de ocupar uma área pequena, a orografia desta Serra proporciona considerável variedade microclimática.

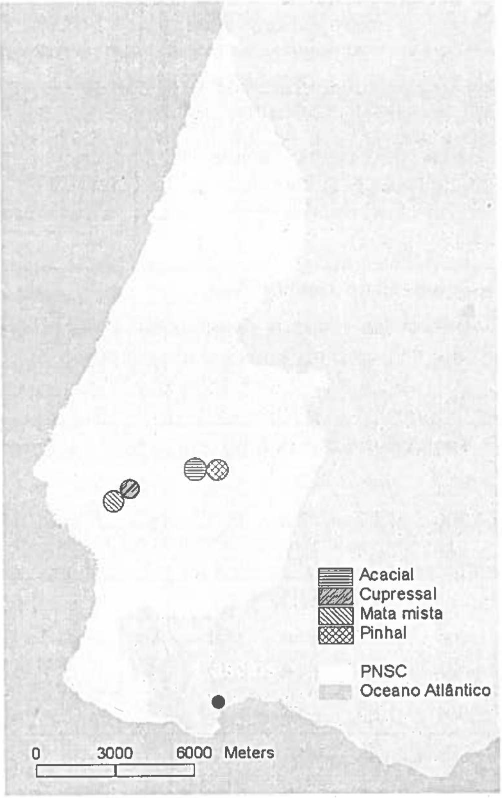

Figura 1. Localização geográfica das matas estudadas.

Na tabela 1 são apresentados os valores mensais das temperaturas mínima, máxima e média do ar, da precipitação e da humidade relativa do ar entre Janeiro de 2000 e Fevereiro de 2001.

O acacial estudado situa-se nos Capuchos (coordenadas UTM 29SMC627927). Trata-se de uma mata muito densa, amplamente dominada por *Acacia longifolia* (Andrews) Willd, onde outras espécies vegetais praticamente não têm expressão. Está exposto a Sudeste, à cota de 350 m. Encontra-se numa zona granítica (RAMALHO *et al*, 1993). O pH do solo é de 5,5.

O cupressal encontra-se entre Vale de Cabeceira e Pedras Irmãs (coordenadas UTM 29SMC602920). É constituído quase exclusivamente por *Cupressus lusitanica* Miller, apesar de também surgirem outras espécies vegetais como *Acacia longifolia* (Andrews) Willd e *Abies alba* Miller.

A densidade de árvores não é tão elevada como no acacial, mas a justaposição das copas leva a um forte ensombramento. Está exposto a Sudoeste e à cota de 350 m. Encontra-se numa zona de sienitos com dioritos e gabros (RAMALHO *et al.*, 1993). O pH do solo é de 5,9.

A mata mista é contígua ao cupressal, tendo uma localização mais a Sudoeste (coordenadas UTM 29SMC599919). É composta essencialmente por *Acacia longifolia* (Andrews) Willd e *Pinus pinaster* Aiton, mas apresenta uma componente importante de fagáceas, como uma pequena mancha de *Quercus pyrenaica* Willd. Esta mata está exposta a Noroeste, situa-se entre as cotas 330 e 350 m e geologicamente é idêntica ao cupressal (RAMALHO *et al*, 1993). O pH do solo é de 5,9.

O pinhal situa-se perto do acacial, em Mosqueiros (coordenadas UTM 29SMC635926). É dominado por *Pinus pinaster* Aiton, mas apresenta uma diversidade vegetal relativamente maior do que as outras zonas de estudo, com espécies como *Quercus lusitanica* L., *Quercus suber* L. e *Arbutus unedo* L.. Está exposto a Noroeste e à cota de 360 m. Encontra-se numa zona granítica (RAMALHO *et al*, 1993). O pH do solo é de 6,0.

Tabela 1. Valores mensais da temperatura mínima, máxima e média do ar, da precipitação e da humidade relativa do ar às 09 h utc entre Janeiro de 2000 e Fevereiro de 2001 da estação meteorológica Lisboa/Gago Coutinho. (Dados do Instituto de Meteorologia).

Temperatura mínima (°C)

Ano	Jan	Fev	Mar	Abr	Mai	Jun	Jul	Ago	Set	Out	Nov	Dez
2000	5,6	10,2	10,7	10,5	13,9	17,0	17,6	18,3	16,8	14,2	11,5	11,6
2001	10,2	9,6										

Temperatura máxima(°C)

Ano	Jan	Fev	Mar	Abr	Mai	Jun	Jul	Ago	Set	Out	Nov	Dez
2000	12,7	17,6	19,7	16,3	22,0	27,3	28,0	28,8	26,9	22,0	16,6	15,9
2001	14,7	16,0										

Temperatura média (°C)

Ano	Jan	Fev	Mar	Abr	Mai	Jun	Jul	Ago	Set	Out	Nov	Dez
2000	9,2	13,9	15,2	13,4	18,0	22,2	22,8	23,6	21,9	18,1	14,1	13,8
2001	12,5	12,8										

Precipitação (mm)

Ano	Jan	Fev	Mar	Abr	Mai	Jun	Jul	Ago	Set	Out	Nov	Dez
2000	19,3	22,0	17,3	154,7	57,0	0,0	4,7	1,9	13,7	27,6	141,4	291,7
2001	195,5	66,6										

Humidade relativa (%)

Ano	Jan	Fev	Mar	Abr	Mai	Jun	Jul	Ago	Set	Out	Nov	Dez
2000	81,0	82,0	70,0	79,0	77,0	63,0	64,0	60,0	70,0	75,0	83,0	87,0
2001	87,0	81,0										

MATERIAL E MÉTODOS

Em cada uma das matas foi percorrido quinzenalmente o mesmo percurso com aproximadamente 3000 m^2 de área, entre Outubro de 2000 e Fevereiro de 2001. Foram também efectuadas incursões nas diferentes matas durante o mesmo período de tempo. Entre 17 de Dezembro e 7 de Janeiro não foram efectuadas saídas de campo devido às más condições atmosféricas.

Todos os macrofungos encontrados foram registados e colheu-se pelo menos um exemplar de cada espécie. Os espécimes colhidos foram identificados e secos. As respectivas excicatas foram depositadas no Arquivo de Fungos Agaricóides do Centro de Micologia da Universidade de Lisboa.

Para a identificação dos espécimes colhidos foi seguida a classificação proposta por HAWKSWORTH *et al* (1995) para as classes e ordens e a de MOSER (1983) para as famílias, géneros e espécies.

Para a determinação do pH do solo foi usado o método standard (RADOJEVIC & BASHKIN, 1999).

Para a determinação da semelhança entre as comunidades macrofúngicas foram usados os coeficientes não variantes de Dice (2 a/2 a+b+c), de Jaccard (a/a+b+c) e de Sørensen (2 a/b+c), onde "a" designa o número de espécies comuns às duas matas comparadas, "b" o número de espécies de uma das matas e "c" o número de espécies da outra mata (LUDWIG & REYNOLDS, 1988, KREBS, 1989). Estes coeficientes variam entre 0 e 1, indicando o valor 0 a ausência de semelhança entre as matas comparadas, e o valor 1 a sua igualdade.

RESULTADOS

Na totalidade foram identificadas 139 espécies nas matas estudadas (tab. 2). A mata que apresentou a maior diversidade foi o acacial, com 59 espécies macrofúngicas (e 1 variedade), seguida do cupressal com 50 espécies, da mata mista com 43 e do pinhal com 31 espécies.

Tabela 2. Espécies encontradas no acacial, cupressal, mata mista e pinhal com respectiva ordem e grupo funcional (s – saprotrófica, m – micorrízica, p – parasita).

Ordem	Espécie	Grupo funcional	Acacial	Cupressal	Mata mista	Pinhal
Agaricales	*Agaricus* sp. a)	s	x			
	Agaricus sp. b)	s	x			
	Agaricus augustus Fr.	s	x			
	Agaricus campestris L.:Fr.	s	x			
	Agaricus hemorrhoidarius Schulzer	s	x			
	Agaricus lanipes (F. H. Moeller & Jul. Scöff.) J. Hlavacek	s	x			

Tabela 2 (cont.)

Ordem	Espécie	Grupo funcional	Acacial	Cupressal	Mata mista	Pinhal
Agaricales	*Agaricus phaeolepidotus* (F.H. Moeller) F.H. Moeller	s	x			
	Agaricus praesclaresquamosus Freeman	s	x		x	
	Agaricus porphyrizon P. D. Orton	s	x		x	
	Agaricus silvaticus Schaff.:Fr.	s	x			
	Agaricus silvicola (Vittad.) Peck	s	x			
	Agaricus xanthoderma Genever	s	x			
	Agaricus xanthoderma var. *griseus* (A. Pearson) Bon & Capp	s	x			
	Agrocybe semiorbicularis (Bull.:Fr.) Fayod	s	x			
	Amanita citrina (Schaeff.:Fr.) Gray	m				x
	Amanita gemmata (Paulet) Bertillon	m				x
	Amanita magnivolvata Aalton	m		x		
	Amanita muscaria (L. Fr.) Hooker	m			x	x
	Amanita rubescens (Pers.:Fr.) Gray	m				x
	Amanita vaginata (Bull.:Fr.) Vittadini	m			x	x
	Armillaria mellea (Vahl.:Fr.) P. Kumm.	p/s		x	x	
	Clitocybe sp. a)	s	x			
	Clitocybe sp. b)	s	x			
	Clitocybe sp. c)	s	x			
	Clitocybe angustissima (Lasch) P. Kumm.	s		x		
	Clitocybe gibba (Pers.:Fr.) P.Kumm.	s		x		
	Clitocybe nebularis (Batsch.:Fr.) P. Kumm.	s	x	x		
	Clitocybe obsoleta (Batsch.:Fr.) Quél.	s	x			
	Clitocybe phaetophtalma (Pers.) Kuyper	s	x	x		
	Clitopilus intermedius Romagn.	s		x		
	Collybia butyracea (Bull.:Fr.) P. Kumm.	s	x			
	Collybia dryophila (Bull.Fr.) P. Kumm.	s			x	
	Collybia hybrida (Kühner & Romagnesi) Svreck & Kubicke	s		x		
	Coprinus sp.	s	x			
	Coprinus atramentarius (Bull.Fr.) Fr.	s		x		

Tabela 2 (cont.)

Ordem	Espécie	Grupo funcional	Acacial	Cupressal	Mata mista	Pinhal
Agaricales	*Coprinus domesticus* (Bolton.:Fr.) Gray	s	x		x	
	Coprinus micaceus (Bull.:Fr.) Fr.	s	x	x	x	
	Cortinarius sp.	m			x	
	Cortinarius hinnuleus Fr.	m			x	
	Crepidotus sp.	s		x		
	Crepidotus crocophyllus (Berk.) Sacc.	s		x		
	Crepidotus mollis (Schaeff.:Fr.) P. Kumm.	s		x		
	Crepidotus variabilis (Pers.:Fr.) P. Kumm.	s			x	
	Cuphophyllus pratensis (Pers.:Fr.) Bon	s		x		
	Entoloma sp. a)	s	x		x	
	Entoloma sp. b)	s	x	x	x	
	Entoloma papillatum (Bres.) Dennis	s		x		
	Entoloma serrulatum (Pers.:Fr.) Hester	s				x
	Gymnopilus penetrans (Fr.:Fr.) Murrill	s		x		x
	Gymnopilus spectabilis (Fr.:Fr.) Smith	s	x			
	Hebeloma sp.	m			x	
	Hebeloma album Peck (ss Bruchet)	m				x
	Hebeloma pallidoluctuosum Gröger & Zchieschang	m		x		
	Hygrocybe calyptriformis (Berk.) Fayod	m/s		x		
	Hygrocybe coccinea (Schaeff.:Fr.) P. Kumm.	m/s		x		
	Hygrocybe conica (Scop.:Fr.) P. Kumm.	s		x		x
	Hygrocybe lepida Arnolds	m/s	x			
	Hygrocybe obrussea (Fr.:Fr.) Wünche	m/s		x		
	Hypholoma capnoides (Fr.:Fr.) P. Kumm.	s			x	
	Hypholoma fasciculare (Huds.:Fr.) P. Kumm.	s	x		x	
	Inocybe sp. a)	m/s				x
	Inocybe sp. b)	m/s			x	
	Laccaria laccata (Scop.:Fr.) Cooke	m/s		x		x
	Laccaria ohiensis (Mont.) Singer	s	x	x	x	x

Tabela 2 (cont.)

Ordem	Espécie	Grupo funcional	Acacial	Cupressal	Mata mista	Pinhal
Agaricales	*Laccaria proxima* (Boud.) Pat.	s	x		x	x
	Lepiota sp.	s		x		
	Lepiota castanea	s		x		
	Lepista inversa (Scop.:Fr.) Pat	s	x	x	x	
	Lepista nuda (Bull.:Fr.) Cooke	s	x	x		
	Leucopaxillus amarus (Alb. & Schwein.:Fr.) Kuhner	s		x		
	Macrocystidia cucumis (Pers.:Fr.) R. Heim	s		x	x	
	Macrolepiota sp.	s	x			
	Macrolepiota konradii (Huijsman ex P. D. Orton) M. M. Moser	s	x			
	Macrolepiota procera (Scop.:Fr.) Singer	s	x			
	Macrolepiota rickenii (Vellen.) Bellú & Lanzoni	s	x			
	Micromphale perforans (Hoffm.:Fr.) Singer	s				x
	Mycena sp. a)	s		x		
	Mycena sp. b)	s		x		
	Mycena epipterygia (Scop.:Fr.) Gray	s				x
	Mycena pura (Person.:Fr.) P. Kumm.	s	x			
	Mycena seynesii Quél.	s	x			
	Ossicaulis lignatilis (Pers.:Fr.) Redhead & Ginns	s	x			
	Pholiota gummosa (Lasch.:Fr.) Singer	s			x	x
	Pluteus cervinus (Schaef.:Fr.) P. Kumm.	s	x			
	Pluteus salicinus (Person.:Fr.) P. Kumm.	s	x		x	
	Psathyrella sp. a)	s		x		
	Psathyrella sp. b)	s			x	
	Psathyrella candoleana (Fr.:Fr.) Maire	s			x	
	Psathyrella hydrophila (Bull. ex Merát) Maire	s	x			
	Psathyrella lacrymabunda (Bull.:Fr.) M. M. Moser	s	x		x	
	Psathyrella prona (Fr.) Gillet	s		x	x	
	Rhodocybe sp.	s	x			
	Rhodophyllus canosericeus J. E. Lange	s	x	x	x	

Tabela 2 (cont.)

Ordem	Espécie	Grupo funcional	Acacial	Cupressal	Mata mista	Pinhal
Agaricales	*Rusgosomyces carneus* (Bull.:Fr.) Bon	s		x		
	Stropharia aurantiaca (Cooke) Imai	s		x	x	
	Tricholoma scalpturatum (Fr.) Quél.	m				x
	Tricholoma sulfureum (Fr.) Quél.	m		x	x	
	Tricholoma ustale (Fr.:Fr.) P. Kumm.	m				x
	Tricholoma viridifucatum Bon	m	x			
	Tricholomataceae a)	s				x
	Tricholomataceae b)	s		x		
	Tricholomopsis rutilans (Schaeff.:Fr.) Singer	s	x	x		
	Tubaria furfuracea(Pers.:Fr.) Gillet	s	x	x	x	
Boletales	*Boletus aereus* Bull.:Fr.	m	x			
	Boletus edulis Bull.:Fr.		x			
	Chroogomphus rutilus (Schaeff.:Fr.) O.K. Mill.	m			x	
	Hygrophoropsis aurantiaca (Wulfen:Fr.) Maire	m	x			x
	Paxillus sp.	?			x	
	Suillus bovinus (L.:Fr.) Kuntze	m	x			x
	Suillus collinitus (Fr.) Kuntze	m				x
	Xerocomus badius (Fr.:Fr.) Gilbert	m	x			
	Xerocomus rubellus (Krombh.) Quél.	m			x	
Russulales	*Lactarius deliciosus* (L.:Fr.) Gray	m			x	x
	Lactarius mitissimus (Fr.:Fr.) Fr.	m				x
	Russula sp. a)	m				x
	Russula sp. b)	m				x
	Russula amara Kucera	m				x
	Russula amethystina Quél.	m	x	x		
	Russula badia Quél.	m				x
	Russula drimeia Cooke	m	x			
	Russula firmula Romagn.	m				x
	Russula foetens Pers.:Fr.	m		x		

Tabela 2 (cont.)

Ordem	Espécie	Grupo funcional	Acacial	Cupressal	Mata mista	Pinhal
Russulales	*Russula heterophylla* (Fr.) Fr.	m	x			
	Russula pectinatoides Peck	m			x	
	Russula rubrocarminea Romagn.	m	x			
	Russula vinosa Lindbad	m			x	
Outras	*Aleuria aurantia* (Pers.:Fr.) Fukel	s			x	
	Cantharellus cibarius Fr.	m		x		
	Cantharellus lutescens (Pers.) Fr.	s	x			x
	Crucibulum laeve (Huds.ex Rehl.) Ka & Lee	s		x	x	
	Geastrum quadrifidum (Pers.) Pers.	s		x		
	Hydnum rufescens Fr.	s			x	
	Helvella elastica Bull.:Fr.	s		x		
	Lycoperdon perlatum Pers.	s			x	
	Peziza sp.	s			x	
	Ramaria sp.	s		x		
	Ramaria sticta (Pers.) Quél.	s	x	x		
	Schizophyllum commune Fr.	s			x	x
	Trichoglossum hirsutum (Pers.:Fr.) Boud.	s		x		

O género mais bem representado foi *Agaricus*, com doze espécies, ocorrendo todas elas no acacial e apenas duas na mata mista. Em todas as matas foram encontradas espécies tidas como nitrófilas (como *Clitocybe nebularis* (Batsch.:Fr.) P. Kumm e *Stropharia aurantiaca* (Cooke) Imai), sugerindo que em todas o solo é rico em azoto. Também em todas as matas se registou a presença de espécies acidófilas (como *Collybia butyracea* (Bull.:Fr.) P. Kumm, *Tricholoma sulfureum* (Fr.) Quél. e *Amanita muscaria* (L.:Fr.) Hooker) o que está de acordo com o pH ácido do solo.

A aplicação dos coeficientes de semelhança aos dados obtidos indica que o conjunto de espécies encontrado nas quatro matas foi substancialmente diferente. As semelhanças resultantes da aplicação dos coeficientes Dice, Jaccard e Sørensen foram concordantes, sendo apresentados apenas as respeitantes ao coeficiente de Sørensen (tab. 3).

Tabela 3. Semelhança entre as diferentes matas de acordo com o coeficiente de Sørensen (diagonal inferior). Na diagonal superior é referido o número de espécies comuns entre as áreas em questão.

	Acacial	Cupressal	Mata mista	Pinhal
Acacial	-	11	12	4
Cupressal	0,21	-	11	4
Mata mista	0,24	0,27	-	7
Pinhal	0,09	0,10	0,12	-

O acacial, o cupressal e a mata mista apresentaram uma clara dominância de fungos saprotróficos, com 82, 78 e 70% da totalidade de espécies respectivamente, enquanto no pinhal os macrofungos micorrízicos foram o grupo dominante (66%).

DISCUSSÃO

O acacial foi a mata que apresentou maior diversidade de espécies macrofúngicas, apesar de ser quase exclusivamente composto por *Acacia longifolia*. Era de esperar que esta maior diversidade fosse encontrada na mata mista, uma vez que esta apresenta maior diversidade de espécies vegetais arbóreas que, à partida, poderiam proporcionar uma maior variedade de condições propícias para o desenvolvimento de diferentes espécies macrofúngicas.

Os valores dos coeficientes de semelhança mostram que as matas estudadas são bastante diferentes entre si, evidenciando que as espécies macrofúngicas têm requisitos ambientais específicos e se encontram contextualizadas em ecossistemas particulares.

FELLNER & PEŠCOVÁ (1995) sugeriram que a predominância de fungos micorrízicos é indicadora de equilíbrio e boa saúde da floresta. Apesar da menor diversidade específica, o facto de no pinhal dominarem os macrofungos micorrízicos sugere que esta será a mata mais equilibrada. Segundo ARNOLDS *et al* (1994), a diversidade de espécies nos fungos micorrízicos é mais complexa do que normalmente é sugerido e, aparentemente, depende da história da floresta, do tipo de gestão e dos efeitos da poluição atmosférica, pelo que são necessários mais estudos para explicar esta questão. De qualquer modo, sabendo que o género *Acacia* não forma ectomicorrizas (ZAK, 1973), é natural que os macrofungos micorrízicos não abundem no acacial (as poucas espécies que surgiram no acacial encontravam-se associadas a espécies vegetais como *Pinus pinaster* Aiton, com representação pontual nesta mata). É importante notar que um aumento na área ocupada pelas espécies do género *Acacia* não augura um bom futuro para os macrofungos micorrízicos.

Apesar de se referirem apenas a quatro meses de trabalho de campo, os resultados apresentados neste trabalho sugerem que os micota das matas da Serra de Sintra apresentam características peculiares e que devem ser estudados com maior profundidade.

AGRADECIMENTOS

À Dra. Fátima Estrela e à Dra. Margarida Barata agradeço o apoio prestado na elaboração deste trabalho.

Este trabalho foi financiado ao abrigo do protocolo estabelecido entre o Instituto da Conservação da Natureza e o Conselho de Reitores das Universidades Portuguesas no ano lectivo de 2000/2001.

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ENVIRONMENTAL IMPACT ON ZOOPLANKTON STRUCTURE OF A SHALLOW ESTUARY (MONDEGO RIVER ESTUARY, PORTUGAL)

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Antunes, C., Rodrigues, E., Pastorinho, M., Bacelar-Nicolau, P., Vieira, L., Azeiteiro, U. & F. Morgado (2002). Environmental impact on zooplankton structure of a shallow estuary (Mondego River Estuary, Portugal). *Revista Biol. (Lisboa)* **20**: 191-207.

The Mondego river estuary is subjected to intense, human disturbance caused essentially by harbour activities, salt-works, aquaculture farms and nutrient and chemical discharge from agriculture areas in the lower river valley. Consequently, and in accordance with recent observations an extensive eutrophication process is taking place in the south arm. In order to have a clear perception of the effective modifications in the ecosystem areas selected from zones affected by human development and zones without pronounced anthropic stress were surveyed to evaluate the human impact on the zooplankton structure. The two sampling stations were located along the southern arm; station 1 is located closest to the mouth of the estuary where a limited impact can be considerable reasonable. Station 2, located in the inner area of the southern arm, in the Pranto river, is severely affected by human activities. The study was carried out from August 1999 through June 2000. The zooplanktonic communities structure was identified by multivariate statistical techniques and species composition and density of the dominant species of each community are compared between the selected areas. Zooplankton samples (obtained by towing a 500 μm mesh net) and associated hydrographic data were collected monthly at the two stations,. Despite the found correspondences with other temperate estuaries, the results showed that in the present zooplankton community spatial structure dominates over seasonal patterns reflecting the environmental stress in this ecosystem.

Key words: Zooplankton structure, environmental impact, Mondego river estuary.

Antunes, C., Rodrigues, E., Pastorinho, M., Bacelar-Nicolau, P.,Vieira, L., Azeiteiro, U. & F. Morgado (2002). Impacto ambiental na estrutura do zooplâncton num estuário pouco profundo (Estuário do Rio Mondego, Portugal). *Revista Biol. (Lisboa)* **20**: 191-207.

O estuário do Rio Mondego encontra-se sujeito a uma intensa perturbação humana essencialmente devida às actividades portuárias, salineiras, de aquacultura e a descargas químicas e de nutrientes provenientes de áreas agrícolas. Em consequência, e de acordo com observações recentes, está a ocorrer um processo extensivo de eutrofização sobretudo no braço Sul do estuário. De modo a ter uma percepção clara das modificações efectivas no ecossistema foram seleccionadas áreas fortemente condicionadas pelo impacto humano e áreas onde esse efeito é menor, de modo a avaliar a influencia ambiental na estrutura do zooplâncton. Foram seleccionadas duas estações de amostragem no braço Sul; a estação 1 foi localizada a próximo da boca do estuário, onde um impacto limitado era esperado. A estação 2 foi localizada na zona a montante, encontrando-se sob forte influência antrópica. O estudo decorreu de Agosto de 1999 até Junho 2000. A estrutura das comunidades zooplanctónicas foi descrita através de técnicas estatísticas multivariadas e, a composição e densidade das espécies dominantes foram comparadas entre as áreas seleccionadas. As recolhas de amostras do zooplâncton (com recurso a uma rede de 500 µm de malha) e a determinação das condições fisico-químicas foram efectuadas mensalmente. Os resultados obtidos mostraram que, para além de semelhanças com outras regiões temperadas, a estrutura das comunidades zooplanctónicas foi dominada pela componente espacial em relação à componente sazonal, reflectindo em parte o stress ambiental do ecossistema.

Palavras chave: Estrutura do zooplâncton, impacto ambiental, estuário do Rio Mondego.

INTRODUCTION

The Mondego river estuary has been subjected, throughout the last decades, to major anthropogenic influences. Among these influences have been important hydrological alterations, population growth, and the development of harbour activities, salt-works, aquaculture farms and nutrient and chemical discharges from agriculture and urban areas in the lower river valley.

Furthermore, due to the semi-closed characteristics of its south arm, poorly connected with the open ocean, and due to it shallowness, the tidal excursion has pronounced delays. Moreover, as a function of the residence time, the persistence of nutrients in the water column, essentially nitrogen and phosphorous, is also relatively long (MARQUES *et al.*, 1993). Consequently, and in accordance to recent observations, an extensive eutrophication process is taking place in this area resulting in seasonal modifications in the specific faunal composition (biodiversity) (MARQUES *et al.*, 1993, 1997; FLINDT *et al.*, 1997; PARDAL, 1998; AZEITEIRO, 1999).

In recent years many studies have assessed the effects of pollutants on different marine organisms (ARNOTT & AHSANULLAH, 1979; CRISAFI, 1974; WARD & BALLANTAINE, 1985; SIOKOU & PAPATHANASSIOU, 1991; SOETAERT & RIJSWIJK, 1993; TELESH *et al.*, 1999; TEDENGREN *et al.*, 1999) and adverse effects may be expected at species composition and trophic structure levels (LILLEBØ *et al.*, 1999). This stresses the need to monitor environmental and biological variables in order to get a clear picture of the effective modifications on the ecosystem structure and function.

In this work two areas were surveyed, one with slight human impact and other notoriously conditioned by human activities. This study aims to evaluate the environmental impact on the zooplankton structure, and the objectives are the evaluation of the variation in the environmental parameters, characterisation of pelagic groups in terms of species composition and density of the dominant species of each community compared between the selected areas.

MATERIAL AND METHODS

Study site

Located in the western coast of Portugal, the Mondego River Estuary (40° 08' N 8° 50 W) possesses a volume of 0.0075 Km^3. The tidal range varies between 0.35 to 3.3 m, being the freshwater discharge of 8.5 x 10^9 $m^3.s^{-1}$.The main area (3.3 Km^2) consists in two arms, north and south.

The south arm is only 2 to 4 m deep, as a consequence of being almost completely silted up in the upstream areas. Water circulation depends basically on tides and, in much smaller amount, the freshwater discharge from a tributary of the Mondego (Pranto River). These facts lead to a residence time of nine days (much longer than the two day period for the north arm) (Fig. 1). During high precipitation periods freshwater runoff may be intense in the south arm due to the opening of sluices that control the water level in the rice fields located upstream. The tide regime may be characterised as mesotidal, varying from 1 to 3 m.

Due to these particular characteristics environmental stress becomes a influential process with the corresponding effects such as euthrophication.

Two sampling stations were selected: Station 1 (S1), in the main body of the estuary with varying depths of 2 to 4 m. Bottom substrata consists of coarse to medium sand, with low organic and carbonate contents and stronger tidal influence (MARQUES *et al.*, 1993). Due to the described characteristics one could argue the existence of limited impact upon this station; Station 2 (S2), in the Pranto River, presents shallow waters, fine sediments rich in organic matter and carbonate contents (MARQUES *et al.*, 1993) and strongly impacted by human activities (MARQUES *et al.*, 1993, 1997; MARTINS *et al.*, 1999; FLINDT *et al.*, 1997; PARDAL, 1998; AZEITEIRO, 1999; LILLEBØ *et al.*, 1999) (Fig. 1).

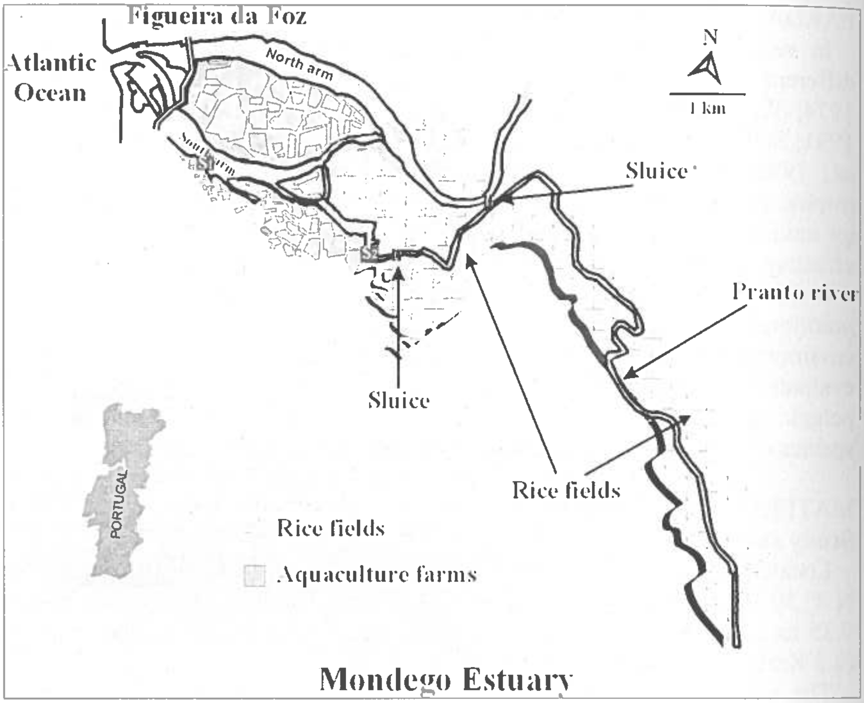

Figure 1. Mondego River estuary. S represents the two sampling stations.

Field sampling

Daytime samples were taken monthly at high spring tides from August 1999 to June 2000. A 500 µm mesh size net (equipped with a HYDOBRIOS flowmeter) was towed for 3 minutes at surface level. Obtained samples were preserved with 4% buffered Formalin.

Measurements of salinity, temperature, dissolved oxygen and pH were taken simultaneously with samples collection. Chlorophyll *a*, nitrite, nitrate and phosphate concentrations were determined from water collected at the stations.

Laboratory procedures

Organisms were identified, counted and densities were calculated per cubic meter (n ind. m^{-3}). Nitrite, nitrate, phosphate, ammonia and chlorophyll *a* concentrations were determined following methods described by STRICKLAND and PARSONS (1968).

Statistical analysis

Physicochemical data analysis

Principal component analysis (PCA; SPAD 3.5, Cisia-Ceresta) was applied in order to identify the major sources of variability in the physical and chemical parameters. Correlation matrices were calculated using the Pearson coefficient.

Communities data analysis

Taxa cluster analysis was performed by the UPGMA method, using Pearson's correlation coefficient (LEGENDRE & LEGENDRE, 1984).

A one-way analysis of variance (ANOVA) was used to test differences of densities between sampling stations and between months.

Zooplankton abundance, number of species and densities of the most abundant species were correlated with salinity, temperature, dissolved oxygen, chlorophyll *a*, pH, nitrites, nitrates and phosphates (ZAR, 1996). Data was log-transformed prior to the ANOVA, in all cases.

RESULTS

Physicochemical parameters and nutrients

The two sampling stations displayed significant different patterns of temporal variation of the environmental variables. S1, under direct marine influence, was characterised by higher values for salinity, DO_2 and pH and lower for temperature. In terms of variability this station was more stable for all parameters except pH. Nitrites and nitrates in S2 registered higher values (Fig. 2).

The PCA of the environmental and biological factors *versus* sampling stations data matrices, considering the first three axes, explained 69.45 % of data variability. The first axis (PC1; 43.89 %) revealed a strong opposition between the two sampling stations, therefore defining a spatial influence: samples characterised by a greater levels of total nitrogen, nitrite, chlorophyll *a*, phosphate and temperature largely corresponded to those from S2, while samples characterised by higher salinity corresponded to those from S1. The second axis of variability (PC2; 14.87 %) opposed samples characterised by higher levels of sulphate and salinity, collected during Autumn 1999 (mainly at S2), and samples characterised by higher levels of nitrate, DO_2 and pH, which correspond to great extent to those collected during the Spring of 2000 (especially May 2000). The third axis of variability (PC3; 10.69 %) reveal an opposition between samples characterised by higher levels of temperature and pH, corresponding to samples mostly collected during Summer, and samples characterised by higher levels of nitrate, corresponding to samples collected during the Winter of 1999 and Spring of 2000. Therefore, the second and third factors defined two distinct temporal patterns (Autumn *vs* Spring; and Winter / Spring *vs* Summer) (Fig. 3).

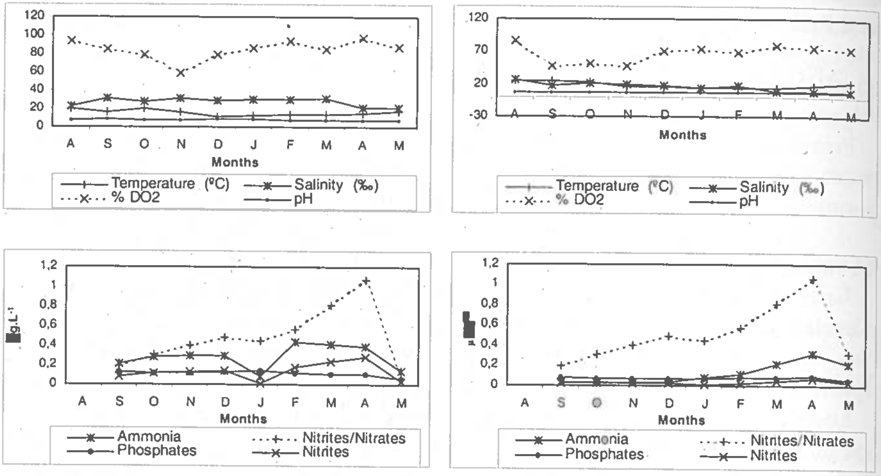

Figure 2. Seasonal variation of physical and chemical variables (temperature, salinity and dissolved oxygen) and nutrients (ammonia, nitrates and phosphates). (A) sampling station 1; (B) sampling station 2.

Quantitative variation of chlorophyll *a* and zooplankton composition

Chlorophyll *a*, in S1, registered a mean of 0.5 μgl^{-1}, with a maximum of 2.7 μgl^{-1} in October 1999 and a minimum of 0.2 μgl^{-1} in January 2000 (Fig. 4 A). In S2, chlorophyll *a* showed consistently higher values, with a mean of 1.7 μgl^{-1}, a maximum of 2.7 μgl^{-1} in May 2000 and a minimum of 0.4 μgL^{-1} in October 1999 (Fig. 4 B).

The zooplankton composition and abundance of dominant species were different between sites as well as between seasons. Holoplankton at station 1, on most occasions, was dominated by Copepoda, but Siphonophora (48 % in August and 10 % in October), Isopoda (15 % in August, 18 % in September and 3 % in April), and Chaetognatha (5 % in November and 12 % in December) were also frequently observed. At station 2, Mysidacea, Copepoda and Isopoda dominate the holoplankton. Within the meroplankton, at station 1, Mollusca larvae and eggs (4 % in November, 2 % in January and 52 % in May), Decapoda larvae (22 % in August and 4 % in April) and Pisces eggs (7 % in March) were frequently observed. At station 2, Mollusca larva and eggs dominate the meroplankton (Fig. 5).

Among the different *taxa* identified in the two sampled areas, the 15 more abundant species were selected for analysis (Tab. I). The dominant species, at station 1, were the post veligers of *Hydrobia ulvae, Acartia clausi, Sagitta friderici, Acartia bifilosa* var. *inermis Muggiaea atlantica, Temora longicornis* and *Paracalanus parvus.* At Station 2, *Mesopodopsis slabberi and Acartia tonsa* were by far the most abundant species followed by *Daphnia* spp. and *Simocephalus* spp.

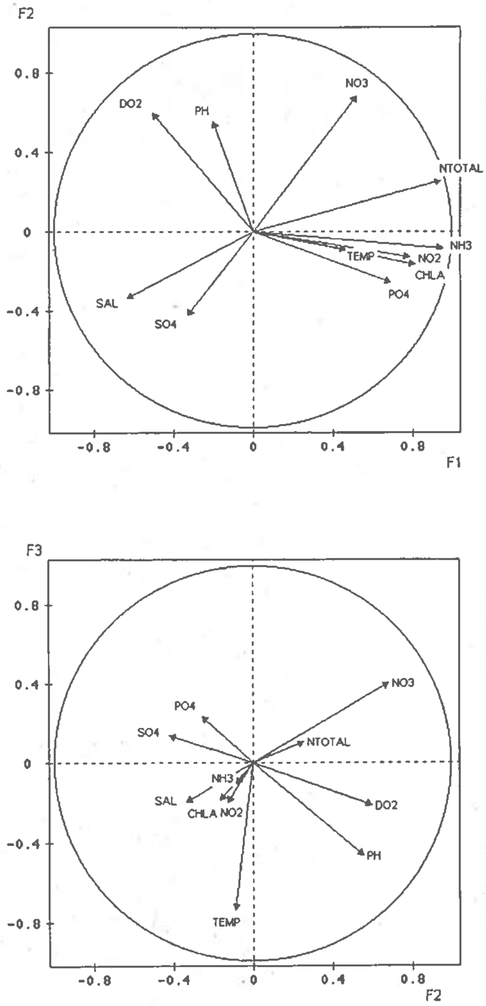

Figure 3. Projection of the principal components F1, F2 and F3 (PC1, PC2 and PC3) for the PCA of physical and chemical factors during the period of the study (SPAD 3.5, Cisia-Ceresta). Descriptors: salinity (SAL), temperature (TEMP), pH (pH), dissolved oxygen (DO2), total nitrogen (NTOT), nitrate (NO3), nitrite (NO2), ammonia (NH3), phosphate (FOSF), sulphate (SO_4) and chlorophyll *a* (CHLA).

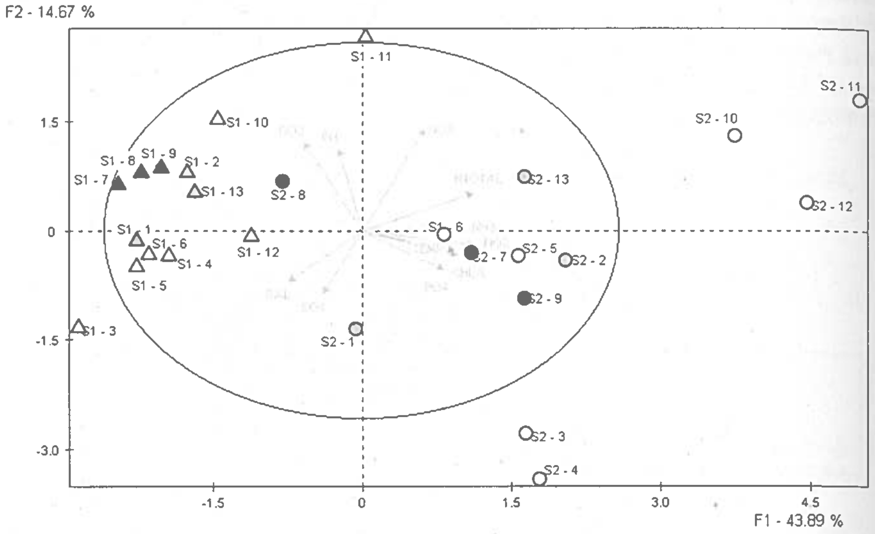

Figure 3 (cont.)

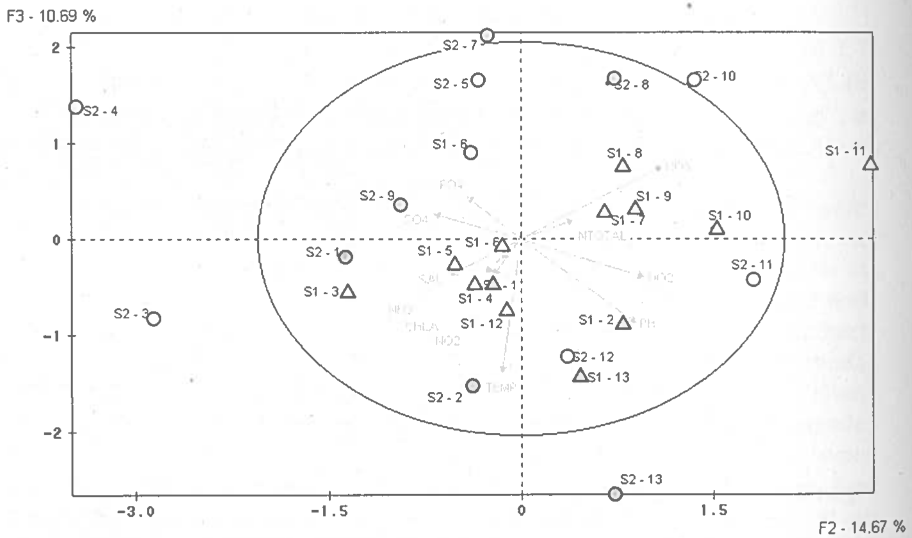

Figure 3 (cont.)

Table I. Mean density and percentage contribution of the most abundant species collected in the Mondego River estuary during the sampling period.

	Mean density	%
Muggiaea atlantica	0,26	1,01
Metatrocofers *Lanice conchilega*	0,13	0,50
Paracalanus parvus	0,22	0,84
Temora longicornis	0,15	0,56
Acartia clausi	2,06	7,94
Acartia bifilosa var. *irnemis*	0,58	2,25
Acartia tonsa	0,71	2,72
Mesopodopsis slabberi	3,50	13,47
Paragnathia formica	0,18	0,70
Daphnia spp.	0,65	2,51
Simocephalus spp.	0,56	2,18
Sagitta friderici	0,45	1,71
Engraulis encrasicolus eggs	0,45	1,75
Pisces eggs	0,36	1,39
Post vel. *Hydrobia ulvae*	13,85	53,35

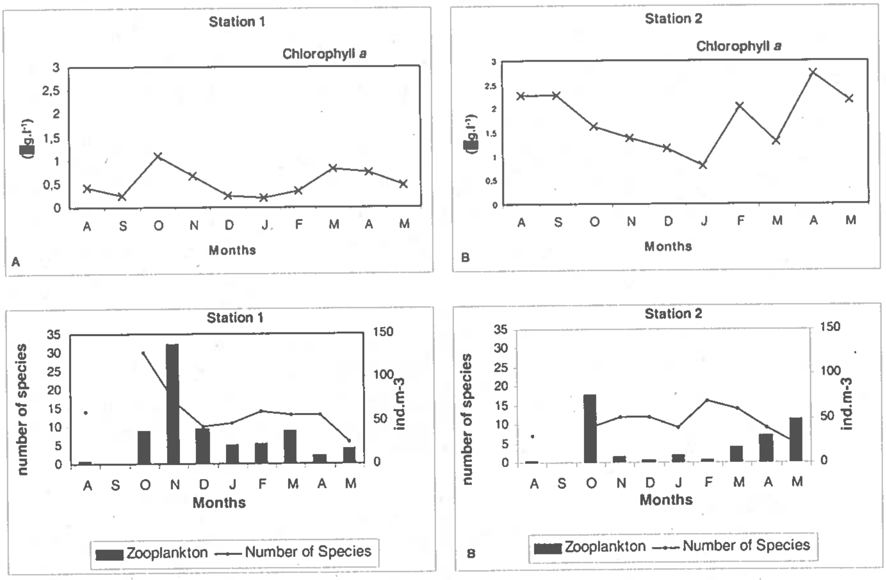

Figure 4 . Seasonal variation of chlorophyll *a*, total zooplankton and number of species. (A) sampling station 1; (B) sampling station 2.

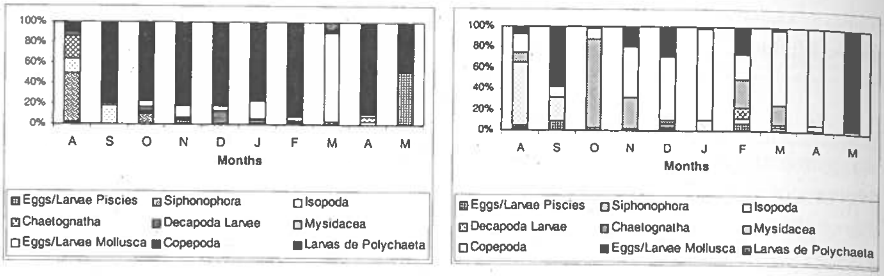

Figure 5. Percentage of the main zooplantonic groups. (A) sampling station 1; (B) sampling station 2

Seasonal and spatial variations

The overall monthly densities of total zooplankton ranged from 1.17 ind.m-3 (August 99; S2) to 137.6 ind.m-3 (November 1999; S1). Number of species ranged from a minimum of 5 taxa (May; S2) to a maximum of 30 taxa (October; S1). Total zooplankton abundance and the number of species were almost always lower during the sampling period in station 2 (Fig. 4). Overall, higher zooplankton densities were found at station 1, with important densities from October to March, reaching a peak in November, with 137.6 ind.m^{-3}. The values of the station 2 were much lower, with three main peaks, in October, with 75.3 ind.m^{-3}, in April, with 30 ind.m^{-3} and May with 48 ind.m^{-3}.

The densities of the most abundant taxa differed amongst months and stations. Although significant differences between months were only registered to *A. clausi* and *A. bifilosa* var. *inermis* ($0.01 < p < 0.001$) and between stations to *A. tonsa* ($p < 0.05$) (Tabs. II & III). The seasonal patterns of abundance of the post veliger of *Hydrobia ulvae* and *Acartia clausi,* in station 1, almost equalled that of the total zooplankton community. Although, at this station, besides the former species, *Muggiaea atlantica* in August, October and November, with, respectively, 1.2 ind.m^{-3}, 2.3 ind.m^{-3} and 1.2 ind.m^{-3}, *Sagitta friderici* in October, with 1.8 ind.m^{-3} and December, with 5 ind.m^{-3}, and *A. bifilosa* var. *inermis* in March, with 5.7 ind.m^{-3} , also showed important densities. The same was observed for the abundance of *Acartia tonsa,* in station 2, followed by *Mesopodopsis slabberi*, with 63.9 ind.m^{-3} in October, *Acartia bifilosa* var. *inermis* with 2.7 ind.m^{-3} in March, post veliger of *Hydrobia* ulvae in May, with 46.7 ind.m^{-3}, and *Daphnia* spp. and *Simocephalus* spp. in January with, respectively, 4.7 ind.m^{-3} and 1.2 ind.m^{-3} and April with, respectively, 8.3 ind.m^{-3} and 10.1 ind.m^{-3} (Fig. 6).

Cluster analysis

Despite the high coincidence of species within the data set, there were detectable differences in community structure between the two selected areas of the estuary, being spatial structure the dominant in the cluster analysis (Fig. 7). A first split (Group A) divides the cluster with the species which occurred with

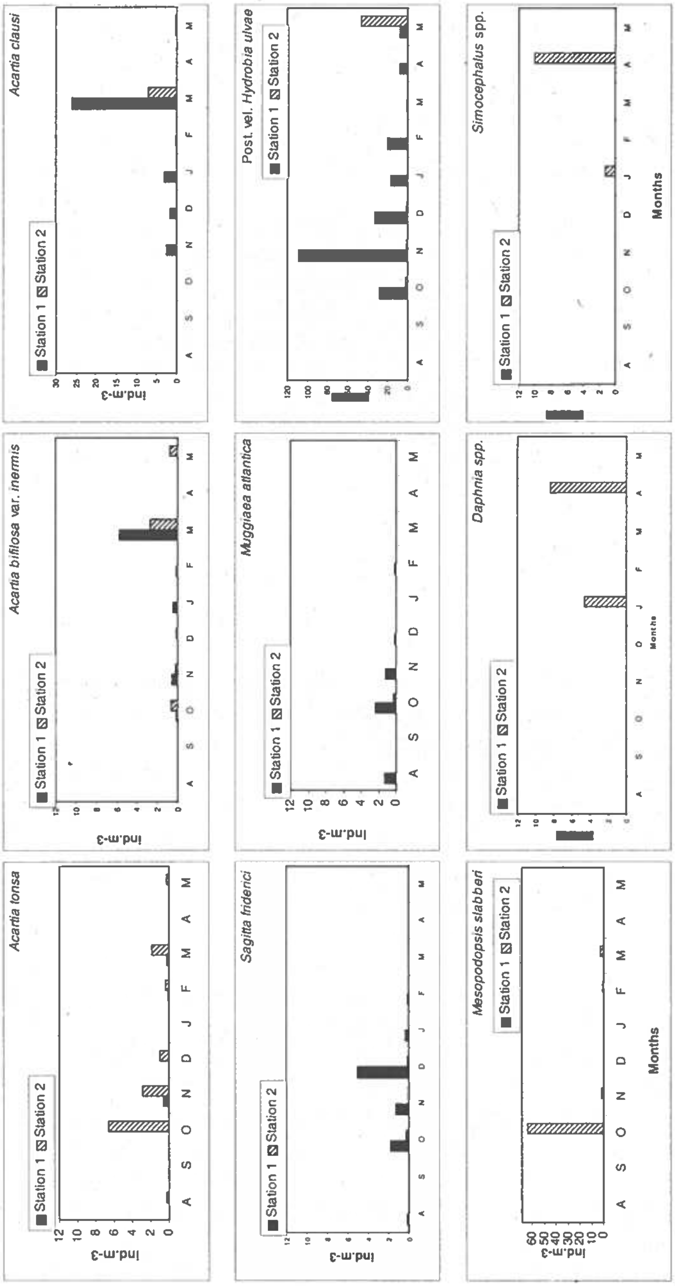

Figure 6. Seasonal variation of the 9 most abundant species density in the sampling stations 1 and 2.

Table II. One-way ANOVA results for the biological parameters at the two stations. The null hypothesis are that, when there are organisms in the water column, their average contribution does not differ across treatments. Df= degrees of freedom; MS= mean square; Fs= test value; P= probability value; P< 0.001- ***; 0.01 > p> 0.001- **; 0.05 > p > 0.01 - *; p > 0.05 - ns.

	STATION				
source of variation	df	MS	F	P	
Total zooplankton	1	0,24433	2,1188	0,16484	ns
Number of species	1	0,24774	2,4305	0,13856	ns
Muggiaea atlantica	1	0,019517	3,78593	0,067497	ns
Metatrocofers *Lanice conchilega*	1	0,01386	0,542856	0,470746	ns
Paracalanus parvus	1	0,025906	1,078501	0,312783	ns
Temora longicornis	1	0,014352	0,744293	0,399638	ns
Acartia clausi	1	0,142818	1,741669	0,203467	ns
Acartia bifilosa var. *inermis*	1	0,04919	0,017284	0,896862	ns
Acartia tonsa	1	0,047873	4,926977	0,05	*
Mesopodopsis slabberi	1	0,162965	3,107001	0,094927	ns
Paragnathia formica	1	0,005258	1,249286	0,278386	ns
Daphnia spp.	1	0,67398	2,206735	0,154713	ns
Simocephalus spp.	1	0,056486	1,707606	0,207746	ns
Sagitta friderici	1	0,035955	3,767067	0,068097	ns
Engraulis encrasicolus eggs	1	0,048987	1,061786	0,316456	ns
Pisces eggs	1	0,034314	1,724514	0,205608	ns
Post vel. *Hydrobia ulvae*	1	0,327455	7,749701	0,01	**

higher densities in the outer station (not impacted), including *Muggiaie atlantica*, *Sagitta friderici*, post veligers of *H. ulvae*, *Paracalanus parvus*. *Temora longicornis* and Pisces eggs. This group can be divided in two sub-groups according to the seasonal occurrence. One sub-group (a1) composed by species which occurred in Autumn and early Winter and a second sub-group (a2) which occurred just in November. A second cluster (Group B) included species which occurred with high abundance in both inner and outer stations (impacted and not impacted), composed by Metatrocofers of *L. conchilega*, *A. clausi*, *A.* bifilosa var. *inermis*, *A. tonsa* and *Mesopodopsis slabberi*. This group can be also divided in two sub-groups according to the seasonal occurrence. A sub-group (b1) included species which occurred since Autumn until early Spring, mainly in station 1; a second sub-group (b2), composed by species which occurred since Autumn until early Spring, mainly in station 2; a third group (Group C), composed by species which occurred mainly in station 2, included *Daphnia* spp., *Simocephalus* spp. and *Paragnathia formica*; this group can be also divided in two sub-groups according to the seasonal occurrence; a sub-group (c1) composed by species which occurred throughout the year and a second sub-group (c2) included species which occurred in Winter and early Spring.

Table III. One-way ANOVA results for the biological parameters during the period of the study. The null hypothesis are that, when there are organisms in the water column, their average contribution does not differ across treatments. Df= degrees of freedom; MS= mean square; Fs= test value; P= probability value; P< 0.001- ***; 0.01 > p> 0.001- **; 0.05 > p > 0.01 - *; p > 0.05 - ns.

	MONTH				
source of variation	df	MS	F	P	
Total zooplankton	9	0,24446	1,1368	0,42269	ns
Number of species	9	0,02095	1,5989	0,24922	ns
Muggiaea atlantica	9	0,0217	1,066026	0,457218	ns
Metatrocofers *Lanice conchilega*	9	0,007528	2,684218	0,069988	ns
Paracalanus parvus	9	0,026146	0,989292	0,501828	ns
Temora longicornis	9	0,012931	1,200505	0,387539	ns
Acartia clausi	9	0,054504	4,63664	0,01	**
Acartia bifilosa var. *inermis*	9	0,009719	9,021211	0,001	***
Acartia tonsa	9	0,057684	1,003063	0,493561	ns
Mesopodopsis slabberi	9	0,195419	0,844631	0,595241	ns
Paragnathia formica	9	0,003146	2,463713	0,088197	ns
Daphnia spp.	9	0,075531	0,892321	0,563172	ns
Simocephalus spp.	9	0,060483	0,933916	0,536199	ns
Sagitta friderici	9	0,041185	1,000332	0,495192	ns
Engraulis encrasicolus eggs	9	0,049576	0,981691	0,506438	ns
Pisces eggs	9	0,037777	0,879607	0,571605	ns
Post vel. *Hydrobia ulvae*	9	0,49875	0,767332	0,649502	ns

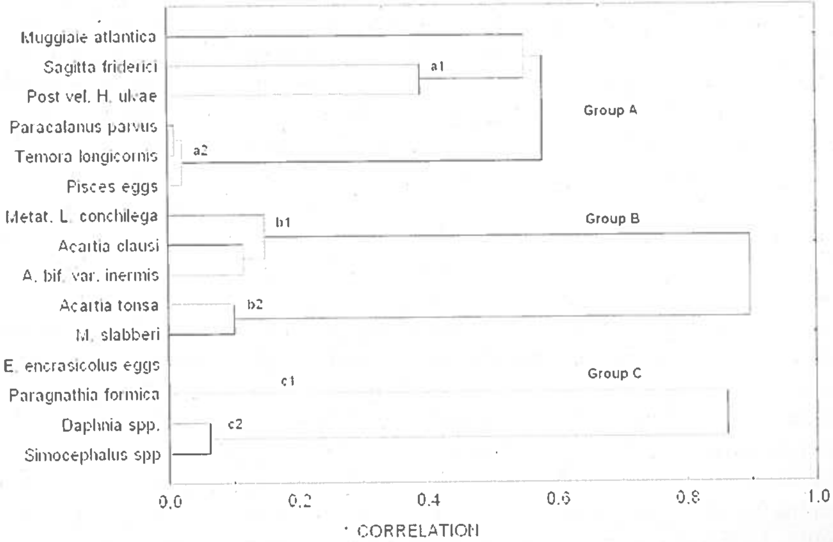

Figure 7. Classification of the most abundant species at both stations during the period of study (clusters A, B and C).

DISCUSSION

Concerning physical and chemical parameters, sampling station 2 presents higher variability and, in general, higher values than station 1 (salinity, temperature amplitudes and more pronounced concentrations of nitrates, nitrites and phosphates, but lower %DO_2). The described patterns are coherent with those described by recent works in the Mondego river (MARQUES et al., 1997, GONÇALVES, 1991, PARDAL, 1998, AZEITEIRO, 1999, AZEITEIRO & MARQUES, 1999).

Salinity values showed some discrepancies towards the expected by registering higher values in winter, January and February, a fact justifiable by the unusual low precipitation rates observed during the year of the sampling campaign. Values for % DO_2 differ from those described by AZEITEIRO (1999), showing higher values during late winter and spring. Phosphate concentration presented very irregular patterns in both stations, but superimposed to this there is an overall tendency to increase concentrations level in late autumn, following the seasonal dynamics already described by AZEITEIRO & MARQUES (1999).

Nutrient values showed seasonal variations related with river discharge, being the most important the nitrogen input, under the form of nitrate (PARDAL, 1998, AZEITEIRO, 1999, AZEITEIRO & MARQUES, 1999). Chlorophyll *a* values were higher in station 2. This fact can be related with the shallower depths and the reduced water circulation, which increases residence time and induces bentho-pelagic mixture, both phenomena contributing to higher chlorophyll *a* concentrations in the upper areas of the estuary.

The results of the PCA analysis of environmental data made possible to recognise a very marked spatial influence (station 1-station 2) and a less marked temporal influence (winter-summer). The study area is shallow (2-4 m during high tide) and almost silted-up, being tides virtually the sole responsible for the water circulation, with the exception of very small freshwater inputs from the Pranto river. Due to these characteristics, this area presents higher sensibility to environmental stress.

The abundance of zooplankton also showed significant spatial and temporal variations during the year, with three main peaks (autumn, early winter and spring). In the Portuguese coast usually the zooplankton production cycle is characterised by the maintenance of relatively high production levels during spring and summer months only diminishing in the winter months (CUNHA, 1993). In this work notable decreases in zooplankton abundances were observed during early Winter and Late Spring and Summer, periods in which the annual minima and richness of zooplankton groups were low related with the disappearance or negligible densities observed for carnivorous zooplankton groups (medusae, siphonophores and chaetognaths). These results reflected some meaningful changes found in the community structure revealed by the short trophic chain and the low taxonomic richness of this area, as well as the absence of carnivorous species.

Results also showed that zooplankton communities at both stations were quite different. Almost all taxa were consistently more abundant in station 1, and a spatial differentiation was revealed by the lower values observed at station 2, both in total zooplankton abundance and in the number of species, which could be the result of the environmental stress which influences this area.

The species composition was typical of many temperate estuaries and of the North Atlantic zooplankton (KENNISH, 1990, FRANSZ *et al.*, 1991) The *Acartia* species dominates zooplankton, followed by the post veliger of *Hydrobia ulvae*, *Mesopodopsis slabberi, Sagitta friderici* and *Muggiaea atlantica.* The densities and the seasonal patterns of abundance of *Acartia clausi* and post veliger of *Hydrobia ulvae* almost equalled that of the total zooplankton community, followed by the high densities of *Mesopodopsis slabberi.* The former species were more abundant in the upstream part of the estuary and the latter in the inner area of the estuary.

Despite the detected similarities to other estuaries of Portugal (MASSAPINA, 1982, SANTOS, 1991, MATTOS, 1995, MORGADO, 1997), the Galicia coast (VALDES *et al.*, 1990), Bay of Biscay (VILLATE, 1991), Westerscheld (SOETAERT & RIJSWIJK, 1993) and Killary harbour (RYAN *et al.*, 1986) the results of this work seem to reflect the environmental stress in this ecosystem, showing that, in the present, zooplankton communities' spatial structure dominates over seasonal patterns.

Results also suggest that the difference in conditions between the two areas were not severe enough to effect significant changes in the stucture of the communities.

The influence of this highly polluted area (southern arm of the Mondego river estuary) upon biological communities has been previously reported for other biological communities (MARQUES *et al.*, 1993, 1997, MARTINS *et al.*, 1999, FLINDT *et al.,* 1997, PARDAL, 1998, AZEITEIRO, 1999, LILLEBØ *et al.*, 1999). These results also illustrate a pattern of environmental stress upon zooplankton composition in this area.

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THE INFLUENCE OF *NEREIS DIVERSICOLOR* ON THE STABILITY AND EROSION OF ESTUARINE COHESIVE SEDIMENTS OF PONTA DA ERVA (TEJO ESTUARY, PORTUGAL)

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Recent studies, have recognized the importance of the biological component for the understanding of the biogeophysical interface processes that rule estuarine sediment transport dynamics. In that context, the effect of the population density of the intertidal polychaete *Nereis diversicolor* on the stability and erosion of estuarine sediments was investigated. Cohesive sediments from Ponta da Erva (Tejo estuary) were collected and stabilized in the laboratory mesocosms with three different densities of *N. diversicolor* (350, 530, 790 ind m^{-2}). After the stabilization period, these were introduced in an annular flume and exposed to shear velocities up to 3.7 cm s^{-1}. Water samples were collected at seven selected velocities (0.7 to 3.7 cm s^{-1}), for suspended and organic particulate matter analysis. Results showed that the density of *N. diversicolor* greatly influences sediment stability increasing erosion. Bioturbation by *N. diversicolor* increases organic matter in the surface sediment. These results confirm the importance of biological processes on the resuspension component of the sedimentary cycle.

Key-words: sediment stability, erosion, *Nereis diversicolor*, annular flume.

Fernandes, S. & Sobral, P. *(2002)*. Influência de *Nereis diversicolor* na estabilidade e erosão dos sedimentos estuarinos da Ponta da Erva (Tejo, Portugal). *Revista Biol. (Lisboa)* **20**: 209-217.

Em estudos recentes, a componente biológica tem ganho uma reconhecida importância na compreensão dos processos de interface biogeofísicos que regem a dinâmica do transporte sedimentar estuarino. Nesse contexto, foi testado o efeito bioturbador do poliqueta *Nereis diversicolor*, uma espécie representativa de zonas estuarinas intertidais. O objectivo deste trabalho foi testar o efeito de *N. diversicolor* na estabilidade e erosão dos sedimentos estuarinos expostos a velocidades de corte até 3,7 cm s^{-1}. Os sedimentos coesivos utilizados foram recolhidos na Ponta da Erva (estuário do Tejo) e estabilizados no laboratório. Foram preparados triplicados com três densidades de *N. diversicolor* (350, 530, 790 ind m^{-2}). Os ensaios de ressuspensão foram efectuados num canal circular, tendo-se recolhido amostras de água em sete velocidades de corte crescentes (de 0,7 a 3,7 cm s^{-1}) para análise de matéria particulada em suspensão e matéria orgânica particulada. Os resultados mostraram que a densidade de *N. diversicolor* influencia a estabilidade e é inversamente proporcional à erosão do sedimento. Foi observado que a actividade de *N. diversicolor* se relaciona com o aumento do teor de matéria orgânica à superfície do sedimento. Estes resultados comprovam a importância dos processos biológicos na componente de ressuspensão do ciclo sedimentar.

Palavras-chave: estabilidade do sedimento, erosão, *Nereis diversicolor*, canal circular.

INTRODUCTION

The net sediment transport from estuaries to the sea can be very small (CANCINO & NEVES, 1999a) and estuaries are known to be trapping zones for sediments and consequentely to associated contaminants (LAANE *et al*, 1999), that is why the study of the processes that influence sediment dynamics on estuarine systems needs to be developed. The processes of sediment transport have been deeply developed on the basis of physical phenomena (BROWN, 1998, DYER & MANNING, 1999, CANCINO & NEVES, 1999 *a* e *b*). Some works focus on the interaction of hydrodynamic and geochemical processes (BLOOM *et al.*, 1992, PUIG & PALANQUES, 1998) but few importance has been given to the biological processes. However, recent studies showed that the effect of biological processes have an influence on sediment behaviour and its response to increasing current velocities and resuspension of particulate matter (MADSEN *et al.*, 1993, BLANCHARD *et al.*, 1997, RUDDY *et al.*, 1998 *a* e *b*, UNDERWOOD *et al.* 1998, WIDDOWS *et al.*, 1998 *b*, WILTSHIRE *et al.* 1998), and should be understood as a functional component that modifies sediment stability and cohesion. Presently there are no adequate models capable of predicting erosion, transport and deposition of fine sediments in estuaries, which is largely because the interactions between physical, chemical and biological processes are poorly quantified and understood (WIDDOWS *et al.*, 2000).

The polychaete *Nereis diversicolor* has a wide geographical distribution from Great Britain to the Mediterranean coasts. In Portugal it is widely distributed, showing regional variations in its reproductive cycle with densities ranging from 16000 to 430 indm^{-2} (FIDALGO E COSTA & CANCELA DA FONSECA, 1998, FIDALGO E COSTA *et al.* 1998, ABRANTES *et al.* 1999). This species is known to have a high tolerance to salinity changes, inhabitating sand, sandy muds, muddy sands and mud (FIDALGO E COSTA & CANCELA DA FONSECA, 1998). Some works show the importance of bioturbating effects of *N. diversicolor* on nutrient regeneration (CLAVERO *et al.* 1991, SVENSSON *et al.* 2000) and on the input of metals to the water column (PETERSEN *et al.* 1998). The aim of this work is to evaluate the effect of three densities of *N. diversicolor* on the stability and erosion of cohesive estuarine sediments exposed to shear velocities up to 3.7 cms^{-1}.

MATERIAL AND METHODS

The effect of *N. diversicolor* on sediment stability and erodability was tested using cohesive sediments from Ponta da Erva (Tejo estuary) collected in November 2000 and March 2001 (organic matter on the sediment was of 7.69% ± 1.63 (mean ± standard deviation). After sediment defaunation, two experiments were performed where mesocosms were prepared with three densities of *N. diversicolor* (450, 600 and 1200 ind m^{-2}) in triplicate, with control mesocosms (CNov and CMar) for each. However, a high mortality was observed on both experiments (average mortality rates of 43%, 69% 25% respectively) and we shall refer each treatment as the average density of each treatment (N350, N530 and N790). Mesocosms were allowed to stabilize under controled continous flow (3 L h^{-1}), temperature, salinity and photoperiod (15 ± 1 °C, 36, 57 ± 11 μ einsteins m^{-2} s^{-1}, 12 h) for a period of 20 days. An annular flume was used and the effect of shear velocity on sediment resuspension was measured. The annular flume (Fig. 1) described in FERNANDES & SOBRAL (2001) and similar to the one described in WIDDOWS *et al.* (1998*a*) has a channel of 15 cm width, with a total area of 0.21 m^2 (outer diameter of 60 cm) and 40 cm height; with a capacity of about 60 L (water column of 30 cm). Current velocities are generated by a rotating drive plate of adjustable height. The drive plate rotation was previously calibrated and may generate shear velocities up to 3.7cm s^{-1}. Water samples were taken at seven selected velocities (from 0.7 to 3.7 cm s^{-1}) and were filtered onto washed, ashed and pre-weighted glass fibre WHATMAN GF/C filters. For suspended particulate matter (SPM) determination the filters were oven dried at 90 °C (48h) and after combustion (450 °C, 1 h) particulate organic matter (POM) was determined.

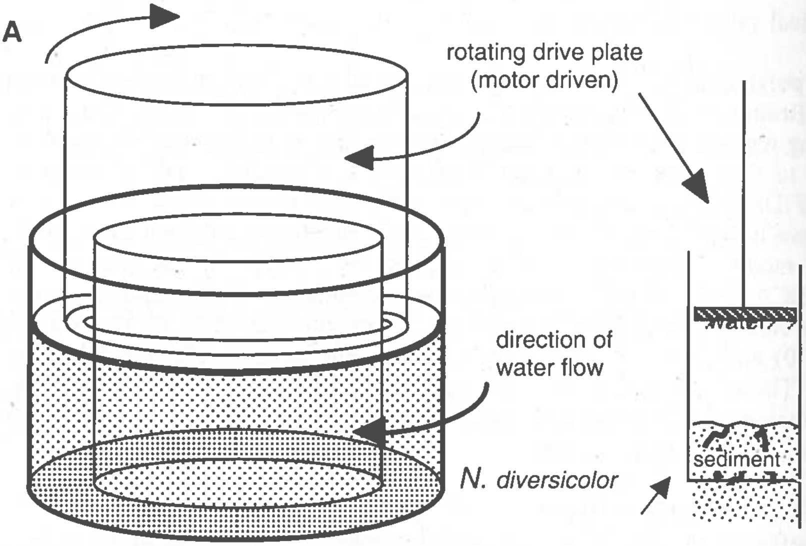


Figure 1. A: Annular flume used to study the effects of bioturbation by *Nereis diversicolor* on the resuspension of sediment exposed to several current velocities. B: Experimental setup (view across the flume channel).

Erosion depth (E_d , cm) was determined using the following relation:

$$E_d = \frac{SPM * V_f}{p_{1cm}}$$

SPM (g L^{-1}) is the concentration of particulate matter at a given velocity, V_f is the volume of water on the flume, and $p_{1\mathrm{cm}}$ is the dry weight of a 1 cm sediment layer (0.75±0.15 g dw, n=13) of the experimental sediment.

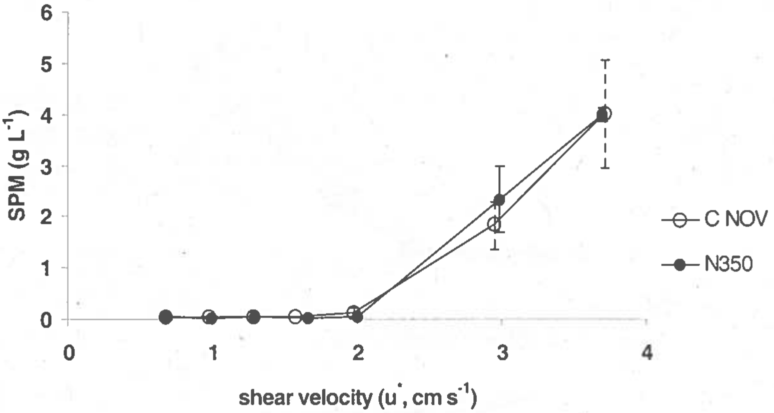
The data were statistically analysed with a multivariate bifactorial ANOVA and with a post-hoc probability test LSD (Least Significant Difference test), using STATISTICA® (version 5.0) software. Independence between samples was assured through the experimental design and variance homogenity was verified through Cochran C, Hartley and Bartlett method after logarithmic transformation of data (ZAR, 1996).

RESULTS

On the experiment with the lowest density no significant differences were found ($p>0.05$) on sediment stability (Fig. 2a) and erodability (Fig. 2b).

SPM concentration at the highest shear velocity (3.7 cms^{-1}) was 3.989 g L^{-1} (± 0.135) in N350 and 4.023 g L^{-1} (±1.057) in the control (C Nov) (Fig. 2a). The same erosion depth of 0.37 cm (±0.02) was found in N350 and 0.37 cm (± 0.24) in the control (C Nov) (Fig. 2b).

2a



2b

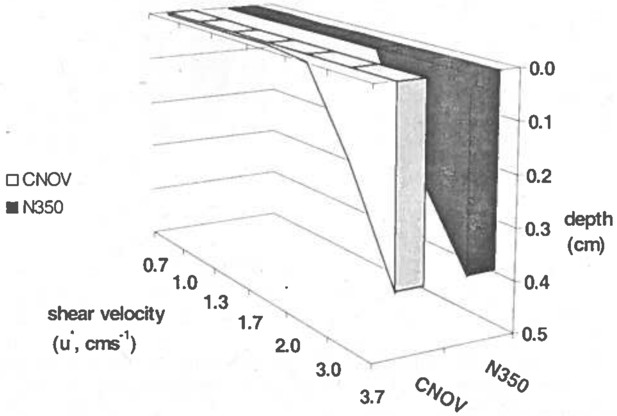


Figure 2. Effect of shear velocity on the resupension of particulate matter (SPM, g L^{-1}) (2a) and on erosion depth (cm) (2b) for the lowest density of *N. diversicolor* (N350) and control (C Nov).

On the second experiment (with the higher densities) statistical analysis revealed significant differences on sediment stability (Fig. 3a) and erodability (Fig. 3b) in N790 and N530 treatments ($p<0.05$). Higher values of SPM were found of 2.682 g L^{-1} (± 0.448) in N790 and 2.405 g L^{-1} (± 0.227) in N530 at 3.7 cms^{-1} while lower concentrations were found in the control (C Mar) 1.923 g L^{-1} (± 0.460). At shear velocities > 2 cm s^{-1} erosion started. Erosion depth at 3.7 cm s^{-1} was 0.18 cm (± 0.04) in the control (C Mar), 0.25 cm (± 0.04) in N790 and 0.22 cm (± 0.02) at N530.

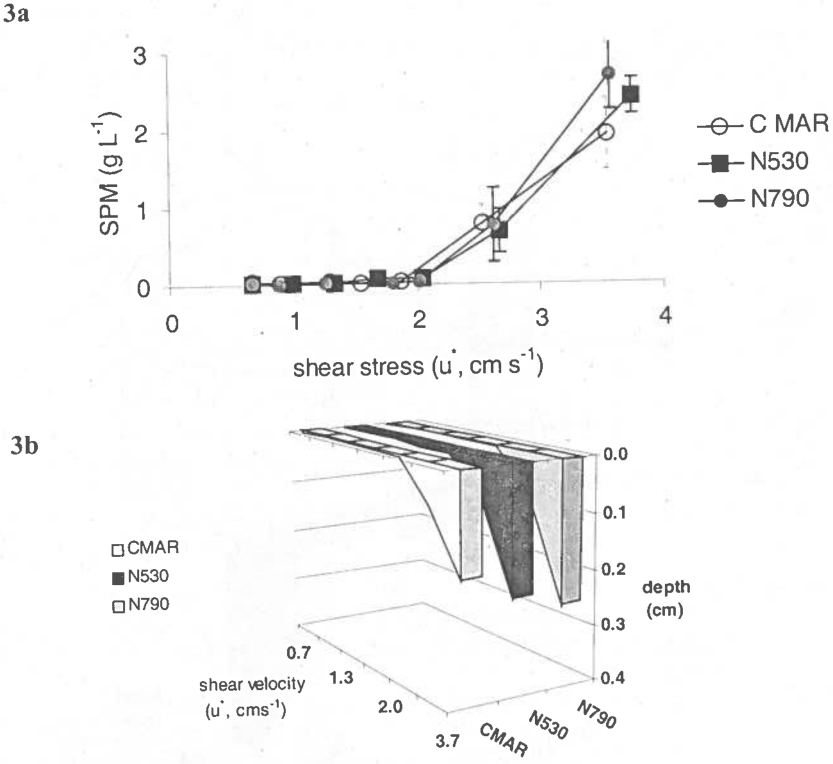


Figure 3. Effect of shear velocity on the resuspension of particulate matter (SPM, g L^{-1}) (3a) and erosion in depth (cm) (3b) for the higher densities of *N. diversicolor* (N790 and N530) and control (C Mar).

Regarding organic matter (%) at the sediment surface, higher values were found on the mesocosms with *N. diversicolor* (Fig. 4). After resuspension takes place and the superficial layer is eroded, organic matter content decreased and was ~ 10 % at maximum shear velocity.

DISCUSSION

The absence of significant differences found on the first experiment is probably due to the low density of *N. diversicolor*, that is considered to be insufficient to produce effect in the sediment stabilisation. However, for the higher densities, the results showed that *N. diversicolor* influencies sediment stability and that density is related to sediment erosion. This destabilizing bioturbation role was already refered in other works (MEADOWS & TAIT, 1989, GRANT & DABURN, 1994) although these authors did not evaluate its influence on sediment stability or erosion. The higher organic matter content found in N790 is probably due to the high mortality observed at the end of stabilisation period that may have enhanced the high organic matter contents found at the sediment surface. Although diferences in sediment stability were found, the bioturbating activity of *N. diversicolor* did not modify the critical shear velocity (2 cm s^{-1}) of the different treatments. The increased organic matter in the sediment is related to *N. diversicolor* activities that seem to be more important at the near surface levels, decreasing with depth.

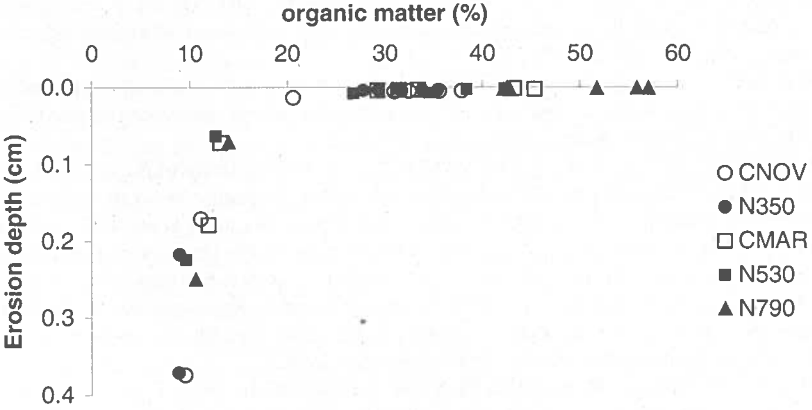


Figure 4. Organic matter content (% MO) and erosion depth (cm) on both experiments with *N. diversicolor* (N350, N530 and N790) and respective controls (C Nov e C Mar).

These results reinforce the importance of biological processes on the resupension component of the sedimentary cycle. Although some authors focus on the bioturbation activity of other macrobenthic organisms (MOURITSEN *et al.*, 1998, WIDDOWS *et al.*, 1998b), the role of the benthic comunity on sediment transport and stability is not fully understood and for that reason, the development of more experiments with an interactive approach between biological and hidrodynamic processes is needed.

ACKNOWLEDGMENTS

This work was supported by a grant (SF) of the Portuguese Foundation for Science and Technology (BTI/17285/98) and by the contract PRAXIS/2/2.1/MAR/1733/95.

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